Climate change and freshwater ecosystems: impacts on water quality and ecological status

Benateau, Simon ; Gaudard, Adrien ; Stamm, Christian ; Altermatt, Florian

Climate change and freshwater ecosystems

IMPACTS ON WATER QUALITY AND ECOLOGICAL STATUS

SIMON BENATEAU, ADRIEN GAUDARD, CHRISTIAN STAMM, AND FLORIAN ALTERMATT

IM AUFTRAG DES BUNDESAMTES FÜR UMWELT BAFU – APRIL 2019

EINE STUDIE IM RAHMEN DES NCCS THEMENSCHWERPUNKTES “HYDROLOGISCHE GRUNDLAGEN ZUM KLIMAWANDEL” DES NATIONAL CENTRE FOR CLIMATE SERVICE

Imprint

Commissioned by: Federal Office for the Environment (FOEN), Hydrology Division, CH-3003 Bern

The FOEN is an agency of the Federal Department of the Environment, Transport, Energy and Communications (DETEC).

Contractor: Eawag, Swiss Federal Institute of Aquatic Science and Technology

Authors: Simon Benateau[1], Adrien Gaudard[2], Christian Stamm[2], Florian Altermatt[1,3]

[1] Department of Aquatic Ecology, Eawag, Swiss Federal Institute of Aquatic Science and Technology, Dübendorf, Switzerland

[2] Department of Environmental Chemistry, Eawag, Swiss Federal Institute of Aquatic Science and Technology, Dübendorf, Switzerland

[3] Department of Evolutionary Biology and Environmental Studies, University of Zürich, Zürich, Switzerland

Corresponding authors: Christian Stamm (Christian.Stamm@eawag.ch) und Florian Altermatt (Florian.Altermatt@eawag.ch)

FOEN support: Fabia Hüsler, Petra Schmocker-Fackel, Ulrich Sieber

Note: This study/report was prepared under contract for the Federal Office for the Environment (FOEN). The contractor bears sole responsibility for the content.

Citation: Benateau S., Gaudard A., Stamm C. & Altermatt F. 2019. Climate change and freshwater ecosystems: Impacts on water quality and ecological status. Hydro-CH2018 Project. Federal Office for the Environment (FOEN), Bern, Switzerland. 110 pp.

DOI: 10.5167/uzh-169641 (https://doi.org/10.5167/uzh-169641)

Foreword

Climate change is one of the largest and possibly most impactful ongoing but also future environmental drivers. Understanding effects of climate change on aquatic ecosystems and their function is thus of high importance. Within the framework of the Hydro-CH2018 project, the Swiss Federal Office for the Environment (FOEN) initiated a synthesis of climate change effects on water quality and the ecological status of freshwater ecosystems. The present report is the result of these synthesis efforts.

Our goal was to summarize available knowledge on how climatic change affects freshwater ecosystems in Switzerland. The work was performed between fall 2017 and fall 2018. We aimed to provide a synthesis of existing knowledge that is as consistent and complete as possible, but also to identify possible knowledge gaps. The work is based on extensive literature surveys and expert interviews. Whenever possible, we included quantitative conclusions. At the same time, we also wanted to be specific and relevant for natural Swiss water systems, so we also included qualitative effects and case studies. While many of the major ongoing and expected effects of climate change on aquatic systems are already well known, we also faced several challenges. For example: (i) some effects of climate change are still only partly understood, (ii) many of the focal endpoints, both with respect to water quality and ecological status, are not only affected by climate change, but also by other anthropogenic drivers, and disentangling these drivers is challenging if not infeasible, (iii) specific effects can be dependent on specific local factors, and (iv) the research field is so vast that the available resources for this one-year synthesis did not allow us to go into as much detail and data analysis as the topic would deserve. As a result, our report is a snapshot of scientific knowledge, at the period of realization, of the effects of climate change on aquatic ecosystems in Switzerland, set into the context of other environmental drivers.

While we see that climate change will have major effects on Swiss water systems, we also identify possible ways of mitigating these effects. We thereby hope that our synthesis is not only an overview of current and future changes, but will also help in policy- and decision-making about how to best deal with these changes and challenges.

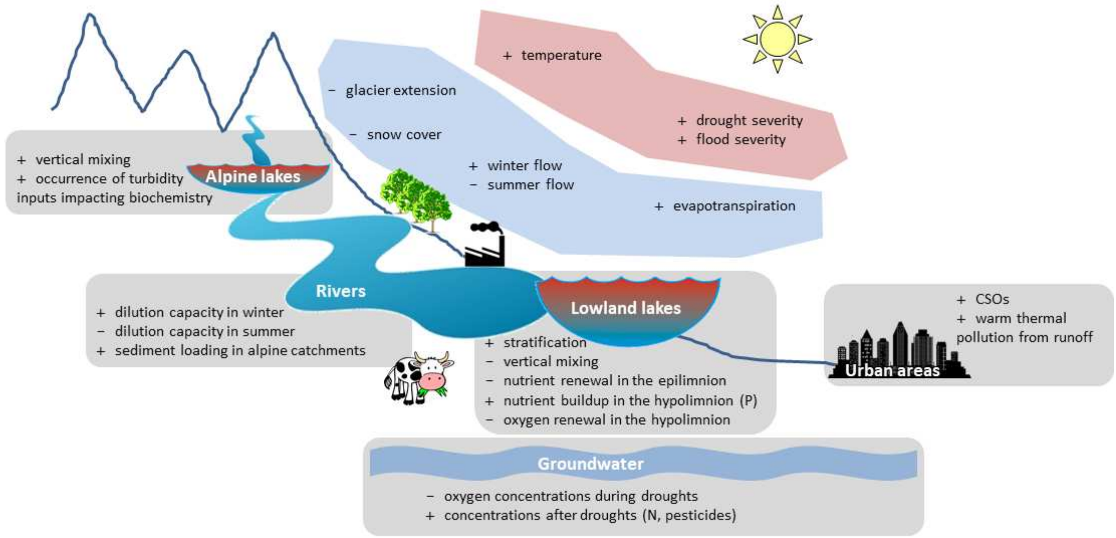
The Authors; Dübendorf, December 2018

Summary and Conclusion

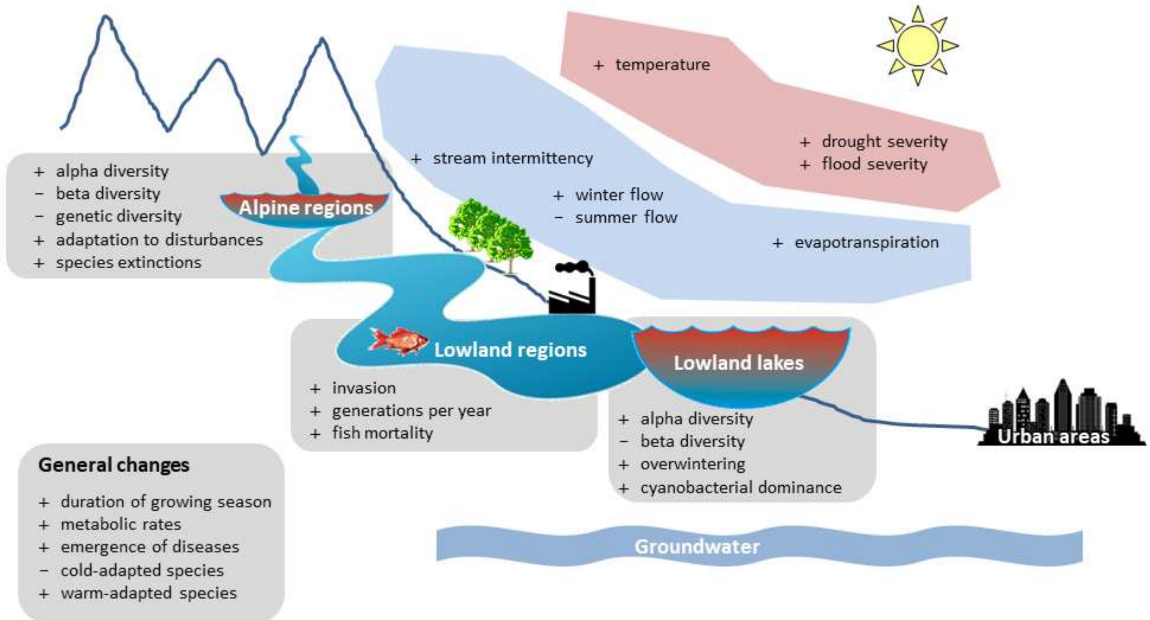
In their Fifth Assessment Report, the Intergovernmental Panel on Climate Change IPCC writes: “the interaction of (i) increased temperature, (ii) increased sediment, nutrient and pollutant loadings from heavy rainfall, (iii) increased concentrations of pollutants during droughts, and (iv) disruption of treatment facilities during floods will reduce raw water quality and pose risks to drinking water quality” (IPCC 2014).

In Switzerland, this statement can be made more specific. Droughts and storms in summer affect transformation and transport of chemicals (typically, during and after rain), especially in urban and agricultural areas. However, over the whole summer, overall chemical loading is likely to be reduced, while in winter, it is expected that soils will be warmer and wetter (especially in the lowlands), favoring biochemical activity and increasing compound mobility. All in all, there are only few impacts on water quality that cannot be prevented through management and adaptation. These include, for example, increasing water temperatures and the seasonal shift of the rivers discharge regime, implying decreasing flows in summer and autumn in non-regulated rivers. Critical impacts to lakes, such as decreasing near-bottom oxygen concentrations and cyanobacterial blooms in late summer, can be partially offset by better nutrient management in their catchments. Similarly, pollution peaks (e.g., combined sewer outflows in urban catchments, or plant protection products in agricultural catchments) can be prevented by promoting reduction of the inputs, and by better runoff and wastewater management. In Switzerland, much has already been done in order to protect the natural aquatic systems and it is to expect that (i) these past and current efforts will allow the retention of good water quality despite climate change, and (ii) further efforts will be useful and necessary, but will be increasingly expensive to put into place. The biological impacts of climate change, however, are much less likely to be prevented or mitigated, such that many of the effects of climate change on the ecological status of aquatic systems discussed in this report will probably happen.

The following figure summarizes the main effects that are expected to occur as direct impacts of warming, and that affect both environmental conditions and the organisms living in freshwater ecosystems in Switzerland. The figure is based on the literature used in this study and can be seen as a visual summary of this work.



(a) Visual synthesis of the impacts of climate change relevant to water quality.
Plus signs (+) indicate increases of a phenomenon, minus signs (-) indicate decreases of a phenomenon.



(b) Visual synthesis of the impacts of climate change relevant to aquatic ecosystems.
Plus signs (+) indicate increases of a phenomenon, minus signs (-) indicate decreases of a phenomenon.

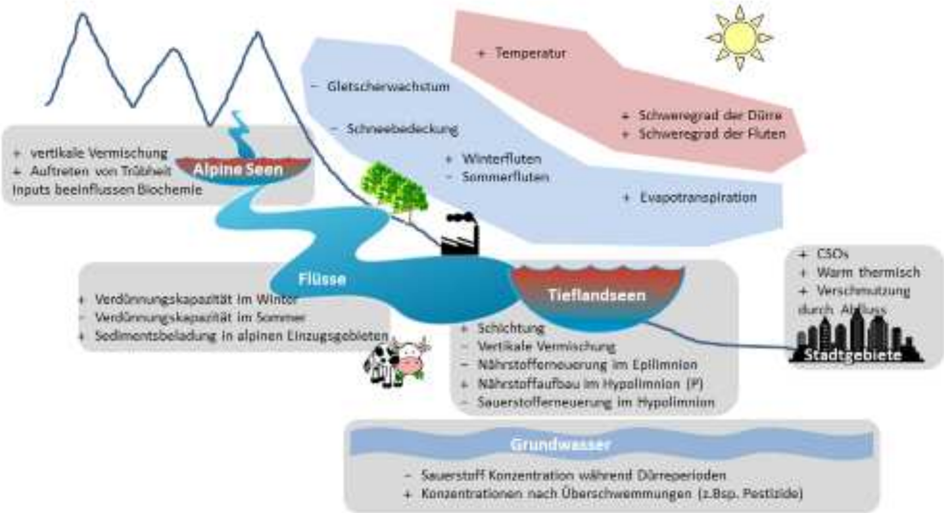
Zusammenfassung und Schlussfolgerung

In ihrem fünften Sachstandsbericht schreibt das Intergovernmental Panel on Climate Change IPCC: „Die Wechselwirkung von (i) erhöhter Temperatur, (ii) erhöhter Sediment-, Nährstoff- und Schadstoffbelastung durch starke Regenfälle, (iii) erhöhten Schadstoffkonzentrationen während Dürreperioden und (iv) Beeinträchtigung von Wasseraufbereitungsanlagen bei Hochwasser werden die natürliche Wasserqualität reduzieren und Risiken für die Trinkwasserqualität darstellen (IPCC 2014).

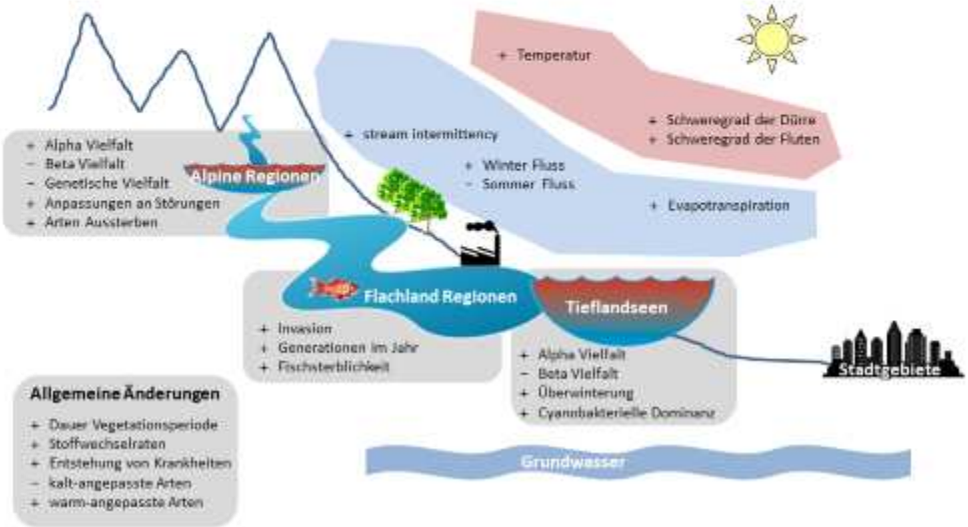
In der Schweiz kann diese Aussage präzisiert werden. Dürren und Stürme im Sommer wirken sich auf die Umwandlung und den Transport von Chemikalien aus (normalerweise während und nach Regenfällen), insbesondere in städtischen und landwirtschaftlichen Gebieten. Während des gesamten Sommers wird die Gesamtbelastung der Chemikalien wahrscheinlich reduziert, während im Winter die Böden wärmer und feuchter sein werden (insbesondere im Tiefland), was die biochemische Aktivität begünstigt und die Mobilität von Chemikalien erhöht.

Insgesamt gibt es nur wenige Auswirkungen auf die Wasserqualität, die nicht durch Management und Anpassung verhindert werden können. Dazu gehören beispielsweise steigende Wassertemperaturen und die saisonale Verschiebung der Abflussregime der Flüsse, was in nicht regulierten Flüssen im Sommer und Herbst eine Abnahme der Abflussmengen bedeutet. Kritische Auswirkungen auf Seen, z. B. die Verringerung der Sauerstoffkonzentration in tieferen Schichten und die Cyanobakterienblüte im Spätsommer, können durch besseres Nährstoffmanagement in den jeweiligen Einzugsgebieten teilweise ausgeglichen werden. In ähnlicher Weise können Verschmutzungsspitzen (z. B. Überlauf von Kläranlagen bei Hochwasserereignissen in städtischen Einzugsgebieten oder Pflanzenschutzmittel in landwirtschaftlichen Einzugsgebieten) verhindert werden, indem die Einträge reduziert werden und die Abwasser- und Abwasserbewirtschaftung verbessert wird. In der Schweiz wurde bereits viel unternommen, um die Gewässer zu schützen, und es ist zu erwarten, dass (i) diese früheren und gegenwärtigen Bemühungen die Erhaltung der guten Wasserqualität trotz des Klimawandels ermöglichen und (ii) weitere Anstrengungen, nützlich und notwendig sind, jedoch auch höhere Kosten verursachen werden. Es ist jedoch weniger wahrscheinlich, dass die biologischen Auswirkungen des Klimawandels verhindert oder abgeschwächt werden, so dass viele der Auswirkungen des Klimawandels auf den ökologischen Zustand der Wassersysteme, die in diesem Bericht erörtert werden, wahrscheinlich eintreten werden.

Die folgende Abbildung fasst die wichtigsten Auswirkungen zusammen, die als direkte Auswirkungen der Erwärmung zu erwarten sind und sowohl die Umweltbedingungen als auch die in Süßwasserökosystemen in der Schweiz lebenden Organismen betreffen. Die Abbildung basiert auf der in dieser Studie genannten Literatur und kann als visuelle Zusammenfassung der Arbeit betrachtet werden.



(a) Visuelle Synthese der Auswirkungen des Klimawandels auf die Wasserqualität.
Pluszeichen (+) zeigen die Zunahme eines Phänomens an, Minuszeichen (-) die Abnahme eines Phänomens an.



(b) Visuelle Synthese der Auswirkungen des Klimawandels für aquatische Ökosysteme.
Pluszeichen (+) zeigen die Zunahme eines Phänomens an, Minuszeichen (-) die Abnahme eines Phänomens

Résumé et Conclusion

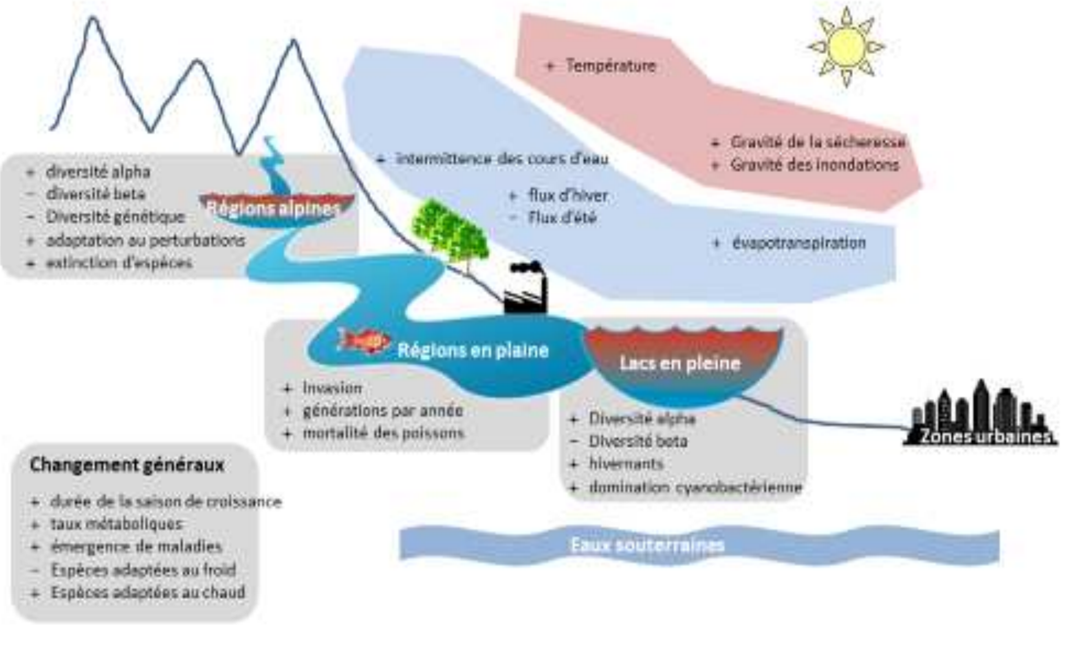
Dans son cinquième rapport d'évaluation, le Intergovernmental Panel on Climate Change IPCC écrit: « l'interaction entre (i) l'augmentation de la température, (ii) l'augmentation de la charge en sédiments, en éléments nutritifs et en polluants due aux fortes précipitations, (iii) l'augmentation des concentrations de polluants pendant les sécheresses et (iv) la perturbation des installations de traitement pendant les inondations réduiront la qualité de l'eau brute et poseront des risques pour la qualité de l'eau potable » (IPCC 2014).

En Suisse, cette affirmation peut être plus précise. Les sécheresses et les tempêtes en été influencent la transformation et le transport des produits chimiques (généralement pendant et après les pluies), en particulier dans les zones urbaines et agricoles. Cependant, au cours de tout l'été, la charge chimique globale devrait être réduite, tandis qu'en hiver, les sols devraient être plus chauds et plus humides (en particulier en plaine), ce qui favoriserait l'activité biochimique et augmenterait la mobilité des composés.

Globalement, il n'y a que peu d'impacts sur la qualité de l'eau qui ne peuvent pas être évités par la gestion et l'adaptation. Ceux-ci incluent, par exemple, l'augmentation de la température de l'eau et le changement saisonnier du régime de décharge des rivières, entraînant une diminution des débits en été et en automne dans les rivières non réglementées. Les impacts critiques sur les lacs, tels que la diminution des concentrations d'oxygène près du fond et la prolifération de cyanobactéries à la fin de l'été, peuvent être compensés partiellement par une meilleure gestion de l'alimentation en nutriments, dans leurs bassins versants. De même, les pics de pollution (par exemple, les déversoirs d'orage dans les bassins urbains ou les produits phytosanitaires dans les bassins agricoles) peuvent être évités en favorisant la réduction des intrants apports et en améliorant la gestion du ruissellement et des eaux usées. En Suisse, beaucoup a déjà été fait pour protéger nos eaux. On peut s'attendre à ce que (i) ces efforts passés et actuels permettent de conserver une eau de bonne qualité malgré le changement climatique, et (ii) des efforts supplémentaires, bien que utiles et nécessaires, seront toujours de plus en plus coûteux à mettre en place. Les impacts biologiques du changement climatique, cependant, sont beaucoup moins susceptibles d'être évités ou atténués, de sorte que de nombreux effets du changement climatique sur l'état écologique des systèmes aquatiques discutés se produiront probablement.

La figure suivante résume les principaux effets attendus du réchauffement sur le climat, qui affectent à la fois les conditions environnementales et les organismes vivant dans les écosystèmes d'eau douce en Suisse. La figure est basée sur la littérature décrite dans cette étude et peut être vue comme un résumé visuel de ce travail.

(a) Synthèse visuelle des impacts du changement climatique sur la qualité de l'eau.
Les signes plus (+) indiquent une augmentation d'un phénomène, les signes moins (-) indiquent une diminution d'un phénomène.



(b) Synthèse visuelle des impacts du changement climatique sur les écosystèmes aquatiques.
Les signes plus (+) indiquent une augmentation d'un phénomène, les signes moins (-) indiquent une diminution d'un phénomène

Table of contents

1 Introduction

Freshwater environments cover less than one percent of the Earth's surface, but nevertheless contain ~10 % of all described species (Dudgeon et al. 2006, Strayer and Dudgeon 2010), making them local and global biodiversity hotspots. They provide numerous ecosystem services (Green et al. 2015) such as water supply for drinking and agricultural use, manufacturing, transport, recreational activities, flood control, and food. However, both globally but also at the scale of Switzerland, aquatic ecosystems are negatively affected by human activities, and biodiversity has been severely reduced (Fischer et al. 2015b). Therefore, climate change is of particular relevance, as it increases the anthropogenic constraint on freshwater ecosystems already facing numerous pressures from human activities (Dodds et al. 2013, Green et al. 2015), such as channelization, dams, irrigation, eutrophication, or pollution.

Climate change is expected to modify the physical characteristics of aquatic environments. Temperature is increasing globally. Precipitation dynamics (magnitude and frequency) are changing, which affects runoff regimes with complex and interacting consequences, including stream intermittency. These climatic changes are particularly important as they will potentially define new environmental conditions, and subsequently affect the status of and processes in freshwater ecosystems (Woodward et al. 2010b).

Much work has been performed to predict future meteorological conditions under expected climate change (IPCC 2014), and their direct environmental impacts in Switzerland (Akademien der Wissenschaften Schweiz 2016). However, the consequences of climate change for water quality and ecological status of freshwater ecosystems were comparatively given less attention. The present work aims at closing these gaps by summarizing knowledge on the direct and indirect processes by which a changing climate affects freshwater ecosystems, with a particular focus on Switzerland.

In this work, we will focus on the effects of climate change through natural pathways, that is, directly related to changing climatic drivers. We will first define these climatic drivers (Section 2), then identify their direct physical impacts on waterbodies (Section 3), highlighting those relevant for freshwater ecosystems. We briefly introduce water quality in the context of land use in Switzerland (Section 4), then review the impacts of climate change on water quality through catchment processes (Section 5) and through processes acting in the waterbodies themselves (Section 6). Finally, the impacts of climate change on ecosystems are reviewed in Sections 7 to 11. Possible indirect effects of climate change through other pathways, for example due to changes in human migration, land-use changes or modified water use, will not be considered but are highlighted in Section 15.1. The structure of the report is shown in Figure 1.1.

This work is based on three main methodological steps:

- **Literature search**. A keyword-based search was performed on Web of Knowledge, and the results were analyzed to reach the most relevant literature and understand the research patterns in the field. This literature search was complemented by integrating literature from related work and from specific searches. More information about the literature review can be found in Section 15.2.
- **Interviews with experts**. Several experts active in the field were selected and interviewed. These interviews allowed efficient acquisition of well-informed knowledge and the latest insights on specific subtopics of this work. In addition, these interviews

provide a valuable assessment of the relative importance of the different processes, as well as a sense of the research gaps. The list of interviewed experts can be found in Section 15.3.

- **Internal reviewing**. Partial or full drafts of the present work were sent to some of the experts for review. The full draft was also reviewed by experts at the FOEN, and all feedback was incorporated in the final version.

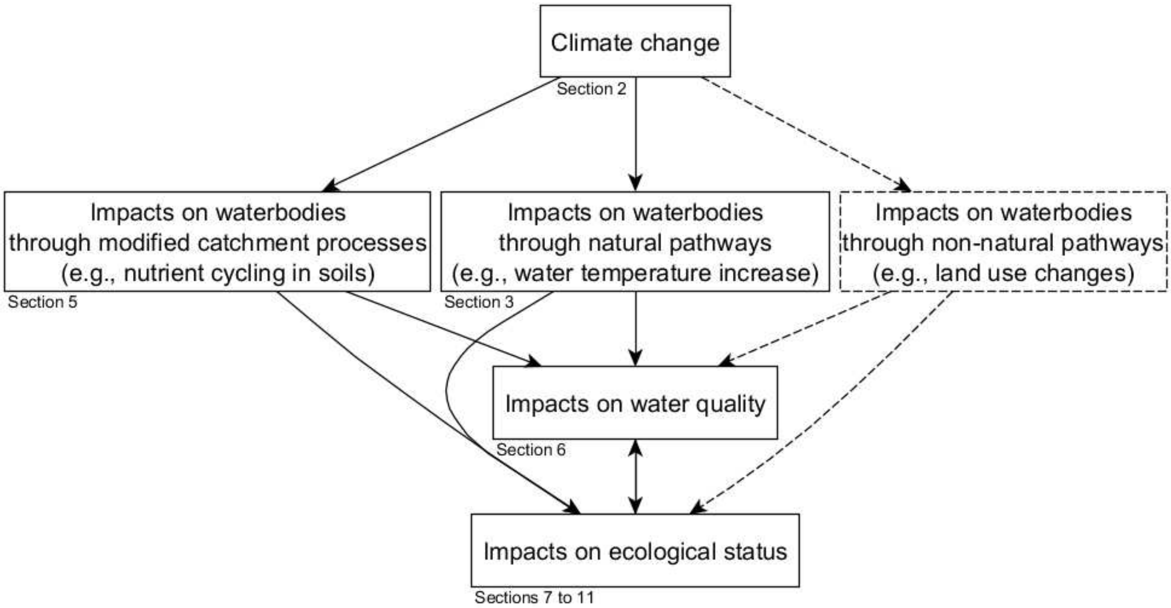


Figure 1.1: Scope and structure of this work.

List of abbreviations

C	Carbon
N	Nitrogen
P	Phosphorus
Si	Silicon
OC	Organic carbon
DOC	Dissolved organic carbon
POC	Particulate organic carbon
TOC	Total organic carbon
DOM	Dissolved organic matter
POM	Particulate organic matter
DON	Dissolved organic nitrogen
POP	Persistent organic pollutant
PCB	Polychlorinated biphenyl
CSO	Combined sewer overflow
PKD	Proliferative kidney disease
FOEN	Federal Office for the Environment
NAWA	National Surface Water Quality Monitoring Programme
NAQUA	National Groundwater Monitoring
BDM	Swiss Biodiversity Monitoring Programme
IPCC	Intergovernmental Panel on Climate Change

2 Climate change: Effects on air temperature and precipitation

2.1 Air temperature

Increasing levels of greenhouse gases in the atmosphere have caused an increase in air temperature, which has been observed over the last decades at global and local levels. Most of the climate-induced warming observed to date has been recorded from air temperature measurements. With current anthropogenic climate change, the mean air temperature in Switzerland is increasing: over the period 1901–2014, annual mean air temperature increased by ~1.9 °C (Scherrer et al. 2016). In the last decades, the intensity of warming has increased with the five warmest years all recorded after 1990 (MeteoSwiss 2018). Over the period 1980–2010, the annual mean temperatures in Switzerland have been increasing at an average rate of 0.35 °C/decade, about 1.6 times faster than the average rate in the Northern hemisphere (CH2011 2011). According to climate scenarios, the annual mean air temperature should increase by 0.9–1.4 °C by 2035 relative to the period 1980–2009 (CH2011 2011). Warming is particularly strong in spring and summer, and is generally more intense in the Alpine region (Rebetez and Reinhard 2008). Climate warming does not happen evenly, but can make breaks and sudden accelerations – the latter was observed in the Northern Hemisphere at the end of the 1980s, with a sudden air temperature increase (+0.9 °C in annual mean), which is believed to be linked to a shift in the Arctic Oscillation (North et al. 2013, Hoffmann et al. 2014).

Climate change has a marked effect on the occurrence of extreme temperatures. In Switzerland, very warm periods (e.g., heat waves) are getting both hotter and more frequent (Scherrer et al. 2016), while cold times (e.g., frost days) are getting rarer (Zubler et al. 2014). These trends are consistent for the whole country. Temperature observations over the 20th century reveal an increase in the frequency of extreme warm events[1] by 5.7 days per decade, together with a reduction of the frequency of extreme cold events[2] by 5.3 days per decade (Figure 2.1; Scherrer et al. 2016).

[1] Daily maximum temperature above the 90th percentile of the study period.
[2] Daily maximum temperature under the 10th percentile of the study period.

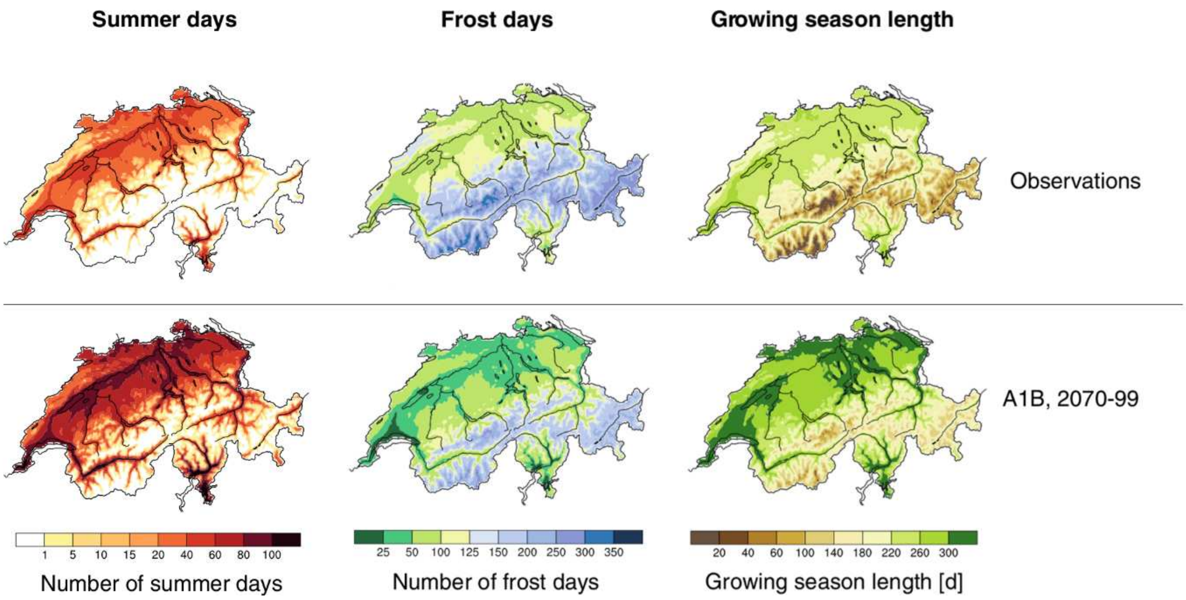


Figure 2.1: Number of summer days (left), number of frost days (center) and length of the growing season (right). Observations: 30-year mean over observations in the reference period 1980–2009, median estimates of RCP3PD-scenario (2070–99) and A1B-scenario (2070–99). The growing season length is given in units of days. Adapted from Zubler et al. (2014).

Depending on the climate scenario, the annual mean number of summer days (max. temperature ≥ 25 °C) on the Swiss Plateau, now about 40 days, is expected to increase to 60–90 days by the end of the 21st century (CH2014, 2014). In the same region and over the same period, the annual mean number of frost days (min. temperature < 0 °C), now about 60 days, is expected to decrease to 30 days (CH2014, 2014). High altitude regions will be most strongly impacted, notably because of the progressive reduction of snow and ice cover (Gurung and Stähli 2014).

Box 1: Limits of climate change predictions.

Quantifying and predicting the magnitude of climate change as well as its subsequent effect on air and water temperature as well as precipitation is difficult, and thus and we must highlight some important limitations. Switzerland has complex topography, necessitating high-quality data and the use of high-resolution models. Local conditions (slope, aspect) also play an important role. Another source of difficulty in making predictions arises from the uncertainty around greenhouse gas emission scenarios. Short-term predictions (until ~2050) are often consistent across scenarios, but for long-term predictions (~2050–2100), the results are strongly affected by the different scenarios' assumptions and by downscaling procedures (Arheimer et al. 2005). In addition, future policies, changes in activities and climate change mediations, and changes in land use are very difficult to predict, and all scenarios must be discussed taking these different contexts into account.

Due to the wide variety of constituent processes, even the newest climate models still have difficulty producing accurate outputs – for instance, they are not yet capable of fully incorporating regime shifts or other complex dynamics, such as a collapse of the Atlantic Ocean overturning circulation ("Gulf Stream"; Caesar et al. 2018). There is good agreement about increasing air temperatures; however, great uncertainties remain in the prediction of precipitation and of its seasonality (Honti et al. 2017). In addition, the future patterns of several other climatic parameters remain unknown (see Section 2.4).

2.2 Precipitation and snow cover

Compared to temperature, precipitation is more variable, and thus more difficult to forecast. Predictions of future precipitation in Switzerland remain rather uncertain (Gurung and Stähli 2014) and, to a certain extent, vary with the climate model used. Three main characteristics of precipitation are expected to change: seasonality, intensity, and rain/snow fraction (FOEN 2012, Fischer et al. 2015a, Morán-Tejeda et al. 2016, Freychet et al. 2017). In the future, and especially in the long term (period 2050–2099), at least the seasonality of precipitation is predicted to change. Summer and autumn are likely to get drier, mainly due to a decreasing number of days with precipitation (i.e., longer sequences of dry days and reduced successions of wet days are expected). By 2085, summer precipitation could decrease by 20–30 % relative to the period 1980–2009, while winter precipitation could increase by up to 20 % (CH2011 2011, Gurung and Stähli 2014). However, this increasing trend is especially marked in southern Switzerland, and less significant in northern Switzerland, where changes are significant only in summer (Figure 2.2; CH2011 2011).

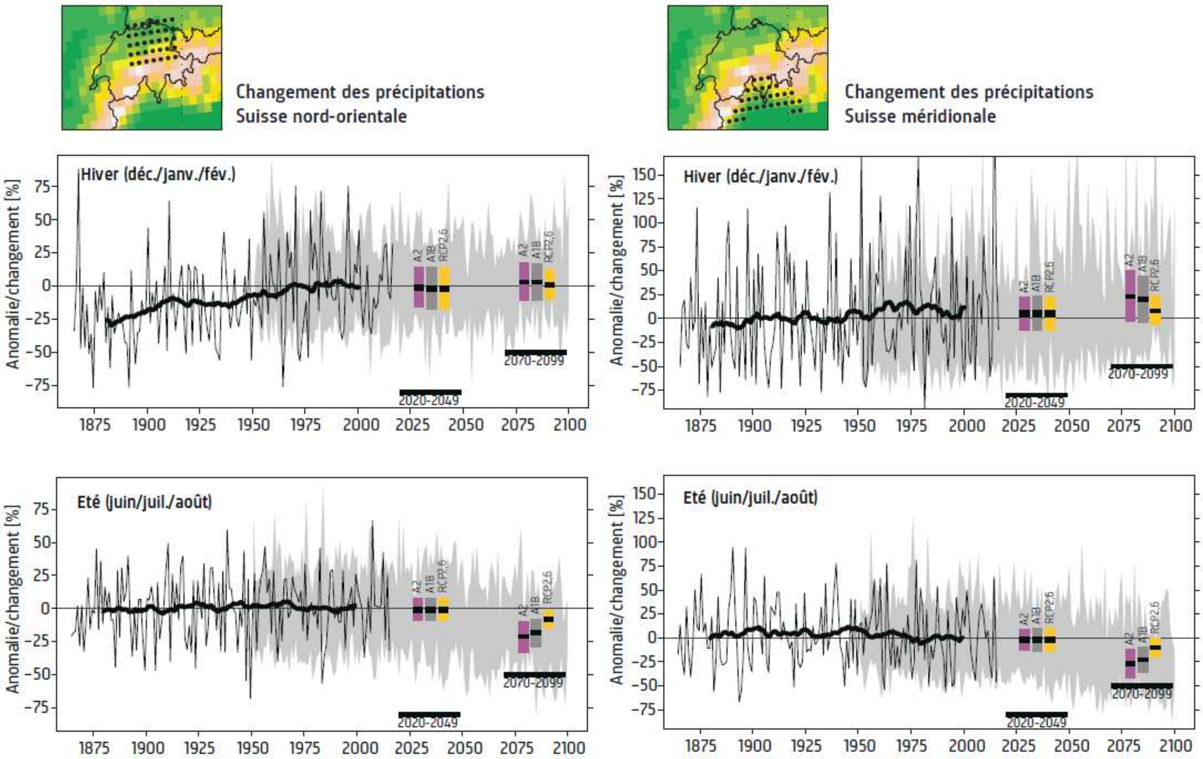


Figure 2.2: Past and actual changes in seasonal precipitation for northeastern Switzerland and southern Switzerland. Changes in percentage relative to the mean of the period 1980–2009. Thin black lines represent the annual observations and thick black lines the mean smoothed over 30 years. Grey areas represent values predicted (5th -95th annual percentile) from the climatic model for the medium emission of greenhouse gas scenario (SRES-A1B). Colored bars represent long-term previsions for three emissions scenarios (high reduction A2, medium reduction A1, low reduction RCP2.6). From CH2011 (2011).

As air temperatures get warmer, an increasing fraction of precipitation will fall as rain instead of snow (Figure 2.3; Morán-Tejeda et al. 2016, Freychet et al. 2017, FOEN 2012). Thus, despite the model uncertainties concerning precipitation under climate change in Switzerland, it is widely accepted that there will be a general increase of rainfall in winter, and a decrease in snowfall (CH2011 2011, Fischer et al. 2015c). When precipitation occurs in form of rain, it contributes directly to runoff, as opposed to the temporal delay inherent to snowfall, when precipitation is stored until the next snowmelt, generally in spring.

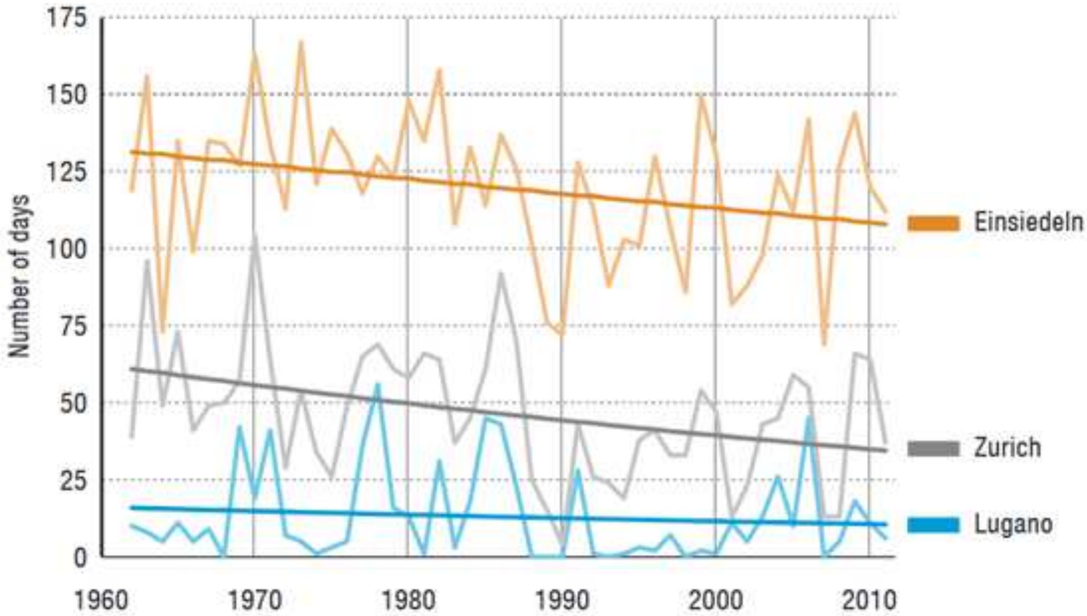


Figure 2.3: Number of days with snow cover. Annual number of days with snow cover and trend lines for the period 1960–2011 in Einsiedeln, Zürich and Lugano. Adapted from Perroud and Bader (2013).

It is also generally expected that the frequency and intensity of extreme precipitation events in Switzerland will increase (Rajczak et al. 2013, Scherrer et al. 2016). This change will presumably be particularly marked in winter, and to a lesser extent in autumn (Gurung and Stähli 2014).

2.3 Snow cover

Snow cover has very high inter-annual variation in the Alps, due to temperature variation and strong fluctuation of the amount of precipitation. It is therefore difficult to predict and even to quantify it (Marty et al. 2017). However, snow cover duration is becoming shorter: Marty et al. (2017) have estimated a reduction of ~9 days per decade in the Swiss Alps. This trend is mainly due to earlier snowmelt and secondarily to a later start of snowfall. The quantity of snow (snow water equivalent) will decrease in the future. The highest impact will be in the regions where the winter temperatures are close to the water melting point, where a reduction of 40–80 % can be expected (FOEN 2012, Steger et al. 2013).

A model for Eastern Switzerland predicted snow cover reductions equivalent to a 800 m elevation shift by the end of the 21st century, with maximal water storage decreasing by up to two thirds and snow season being up to nine weeks shorter (Bavay et al. 2013). Reduced snowpack and briefer snow cover increases glaciers' exposure to the atmosphere, and thus enhances glacial melting. However, the same drivers result in reduced surface insulation and therefore better preservation of permafrost during the cold season. For the soil, snow can be seen as a protection against atmospheric heat fluxes, as the temperature of snow-covered soil is decoupled from air temperature and solar radiation. If conditions change so that snow cover is diminished (e.g., drier conditions or more precipitation as rain) or so that snowpack develops later, then the insulating capacity of snow is reduced and soil freezing can become more frequent (e.g., more diurnal freeze-thaw cycles) and extensive (Edwards et al. 2007).

2.4 Other meteorological parameters

In addition to air temperature and precipitation, other climatic drivers may also change in the future, however their evolution is uncertain:

- **Solar radiation**. Solar radiation is variable, following cycles that are independent of the Earth's climate. Nevertheless, the solar radiation reaching and warming the Earth's lower atmosphere is influenced by cloud cover, while air transparency determines the fraction hitting the surface (Schmid and Köster 2016). In addition, albedo (high for urban areas, low for snow) determines the absorption of sunlight by the Earth's surface.
- **Cloud cover**. As mentioned above, cloud cover determines the amount of solar radiation reaching the lower atmosphere. Increases (resp. decreases) in average cloud cover could have a very strong cooling (resp. warming) effect on the Earth's climate.
- **Relative humidity**. Relative humidity influences many environmental processes such as evaporation, which is of primary importance for surface freshwater (Schindler 1997).
- **Wind**. The strength and regime of wind affects, among others, snow deposition (Campbell et al. 2000) and energy input to surface waters (Rouse et al. 1997). However, wind predictions are rather unreliable, particularly in the alpine range (Etter et al. 2017)

3 Physical impacts of climate change on waterbodies

The above-described direct effects of climate change on air temperature and precipitation have cascading effects on the physical conditions in aquatic habitats, which we discuss in the following subsections separately for lakes, rivers, and groundwater. Early research already highlighted the fact that changes in hydrological variability (frequency and magnitude of extremes) may have a greater potential to impact water resources in many regions than changes in mean annual conditions (Gleick 1989). Here, we will consider both.

Box 2: Impact of climate change on glaciers and consequences for surface freshwater.

In 2010, glaciers covered 944 km^2, corresponding to 2.3 % of the total area of Switzerland (Fischer et al. 2014). This area is shrinking: glaciers have been receding since the end of the 19^{th} century (FOEN 2012, Pellicciotti et al. 2014), and from 1973 to 2010, their area was reduced by 28 % (-363 km^2) (Fischer et al. 2014). According to long-term predictions, a large proportion (50–90 %) of the iced area of Switzerland will disappear by the end of the 21^{st} century (Jouvet et al. 2011, Linsbauer et al. 2013). Even in low-emission climate scenarios, glacier loss is expected to be 50–75 % (Figure 3.1; Salzmann et al. 2012). Air temperature is the main factor currently influencing glacier mass and causing melting. Other influencing environmental variables include solar radiation, wind patterns, and relative humidity (Rahman et al. 2015).

Glaciers are an important source of water for mountainous streams. Although most of the annual runoff originates from snowmelt and rain, glacier meltwater has a strong impact on seasonality as it is concentrated in the snow-free period between July and October (Stahl et al. 2016). As reviewed by Pellicciotti et al. (2014), most studies suggest an increase of the runoff in glacier-fed streams during the next decades as glaciers melt more quickly, followed by decreasing runoff and a shift in seasonality after glacier mass becomes insufficient to maintain the runoff.

The retreat of glaciers leaves space for new rivers and lakes. The movement of ice masses also affects soil pressure and compaction, and therefore the distribution of groundwater in the affected high alpine areas (Gurung and Stähli 2014).

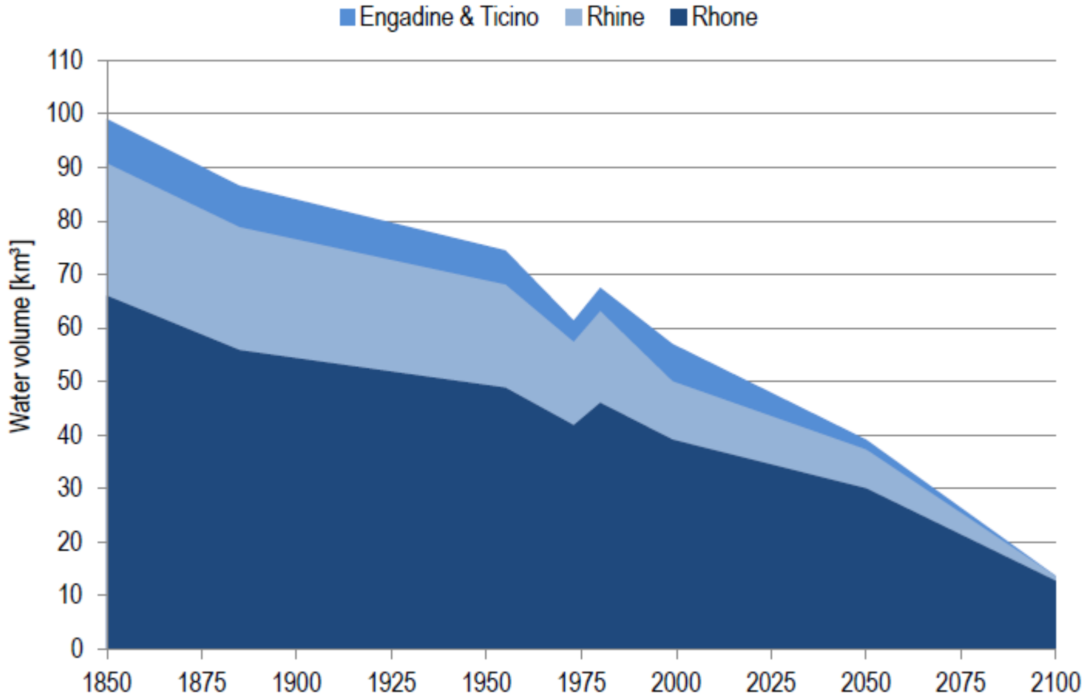


Figure 3.1: Development in volume of water stored in Swiss glaciers (Rhône and Rhine basins, Engadin and Ticino) since the end of the Little Ice Age. Estimated since the end of the Little Ice Age (degree of uncertainty 20–30 %) and simulated up to 2100. From FOEN (2012).

3.1 Rivers

3.1.1 Increase of water temperature

Air temperature strongly influences the temperature of streams and rivers, which are generally less buffered than lakes (Mohseni and Stefan 1999, Meier et al. 2003). This influence is larger (i) far from the river source, (ii) if the river is shallow or slow-flowing, and (iii) if there is little riparian forest to provide shade and buffer daily temperature variation (Caissie 2006). Responding to climatic forcing, rivers have already warmed up and will continue to do so in the 21st century: temperatures of Swiss rivers were observed to increase by 0.1–1.2 °C between 1970 and 2010 (Figure 3.2; FOEN 2012). Rivers with a long course in the lowlands, as well as rivers downstream of lowland lakes, are already particularly warm and will get warmer. On the contrary, in alpine rivers close to their sources, temperature increases will be moderate (FOEN 2012) because snow- and icemelt will remain the main water sources, and external drivers (air temperature, solar radiation, hyporheic exchange, anthropogenic impact, etc.) have a relatively weak influence on water temperature directly downstream of the melting source. Figure 3.2 illustrates the different responses of river temperature to the hot summer of 2003: rivers below lakes showed the strongest response, whereas the Rhône, fed by high alpine catchments, was less affected.

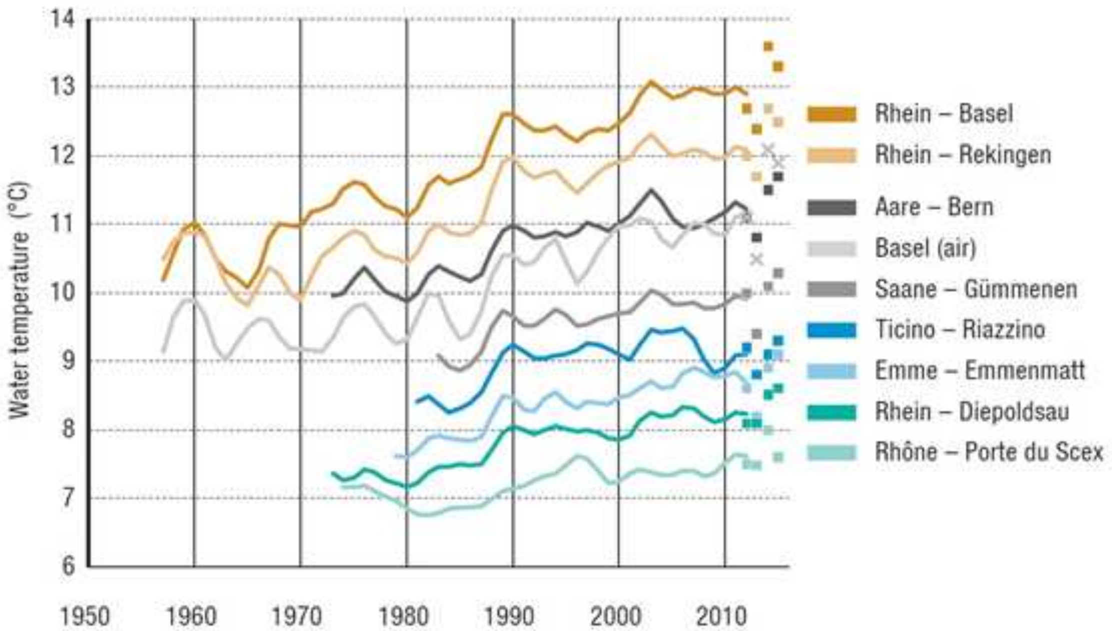


Figure 3.2: Time series of annual mean water temperature in several Swiss rivers. Adapted from Perroud and Bader (2013).

3.1.2 Altered flow regime, reduced ice/snow cover

The main effects of climate change on rivers are (i) altered partitioning between the runoff sources, and (ii) altered flow regimes. River runoff in Switzerland is generated by rain (35 %), snowmelt (40 %), icemelt (2 %), and inflow from abroad (23 %) (FOEN 2012). However these fractions vary depending on the catchment – for example, in the Rhine at Basel, the contribution of rain is higher and in the upper Aare (upstream of Lake Brienz), snowmelt (55 %) and icemelt (13 %) account for more runoff than rain (32 %) (Figure 3.3; Stahl et al. 2016).

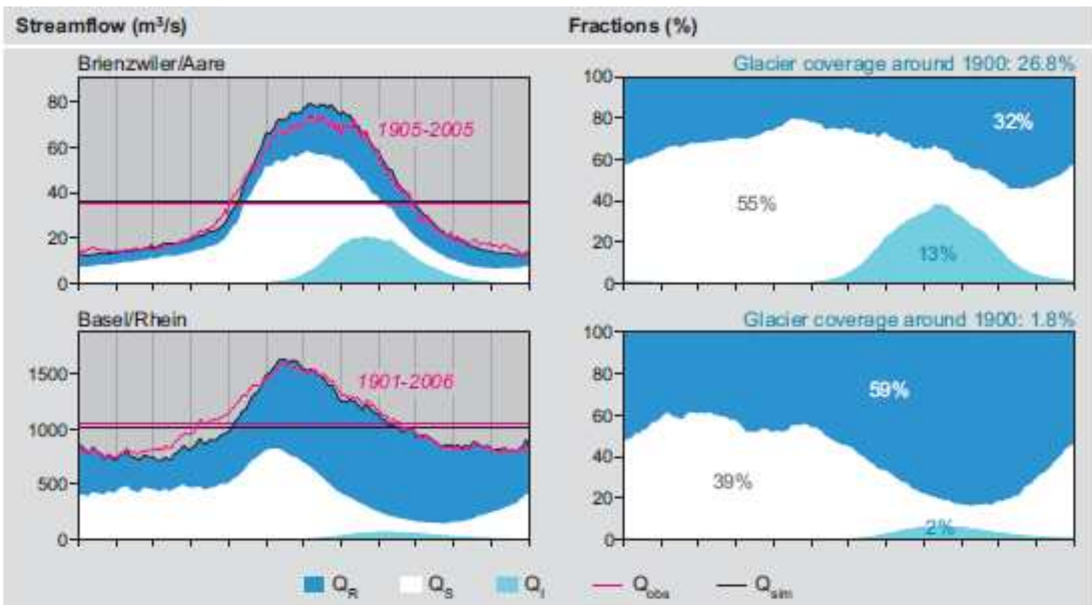


Figure 3.3: Hydrological regimes with modelled streamflow components of rain, snowmelt, and icemelt at selected gauges in the Rhine basin in Switzerland (left: absolute values with long-term mean as a line; right:

relative fractions of the streamflow components with snowmelt component of streamflow in white, rainfall component of streamflow in blue and glacier icemelt component of streamflow in turquoise). Adapted from Stahl et al. (2016).

The high variability of river runoff regimes (see Figure 3.4) makes prediction challenging (Lepori et al. 2015). However, it is generally expected that the seasonal distribution of runoff will change in Switzerland. Indeed, the runoff regime in non-glacial catchments is shifting from nival (snow-controlled) to pluvial (rain-controlled) (Figure 3.4; CH2014-Impacts 2014). In rivers of the Plateau and alpine foothills, which are mainly fed by snowmelt and rain, runoff will increase markedly in winter (due to increased rain), but decrease in summer (due to less snowmelt) (Bouraoui et al. 2004, FOEN 2012, CH2014-Impacts 2014).

As a result, low-water events are likely to occur in summer in most rivers, including those where the minimum flowrates are now reached in winter (FOEN 2012). Summer droughts will presumably get worse in rivers that already experience low summer flow, while some groundwater-fed rivers may be spared (Andersen et al. 2006). The drought of 2003 strongly impacted catchments without large lakes and with minimal discharge from snowmelt (Bader et al. 2004), suggesting that lake regulation may prevent low flows in downstream rivers (Bader et al. 2004).

For many rivers of the Plateau and alpine foothills, flooding will likely occur earlier in the year (i.e., in the second half of winter instead of early summer), responding to both modified rainfall regime and earlier snowmelt (FOEN 2012). It is however worth noting that the occurrence of floods is linked to the North Atlantic Oscillation, and not only to regional climate (Schmocker-Fackel and Naef 2010), thus any change in the North Atlantic Oscillation (Caesar et al. 2018) will have potentially large and unpredictable effects. A tendency towards stronger floods due to climate warming has also been highlighted (FOEN 2012), although uncertainty of this effect on flood magnitude and frequency remains. Expected changes in runoff by region and season were graphically summarized in CH2014-Impacts (2014), as shown in Figure 3.5.

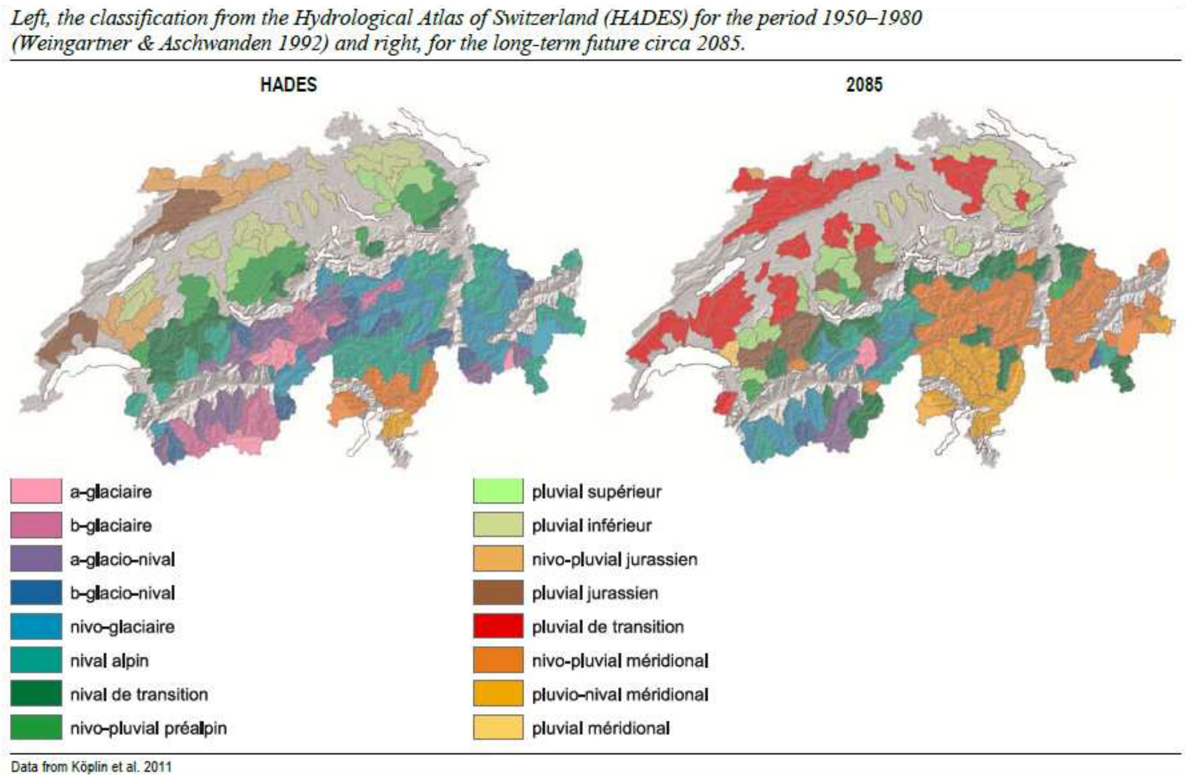


Figure 3.4: Runoff regimes in Switzerland. From (FOEN 2012).

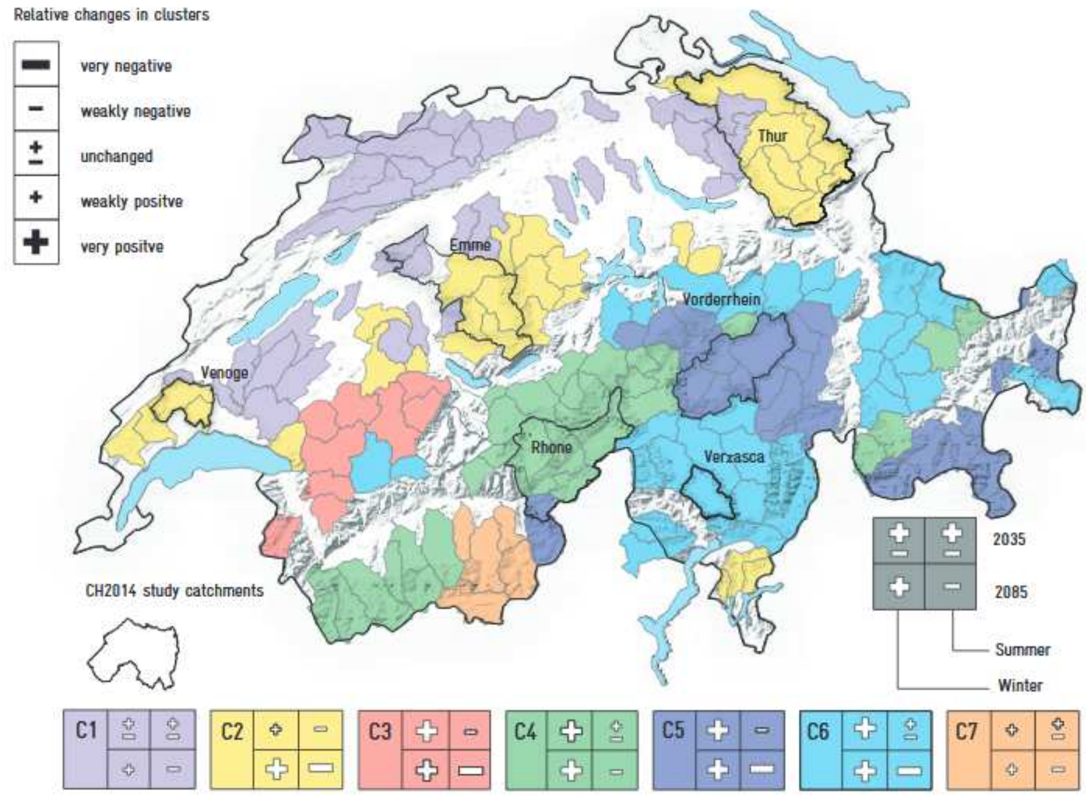


Figure 3.5: Regions where similar seasonal runoff changes resulting from the climate change are expected. Mean values are considered for two periods: 2020–2049 and 2070–2099. Calculations are based on the SRES-A1B scenario of greenhouse gas emissions. In all regions, runoff will increase in winter and decrease in summer. The timing and the size of the change, however, will vary according to the region. The alpine region is particularly affected (blue and green tones), except very high-altitude regions, while the Plateau, Jura and the alpine foothills are less affected (purple and orange tones). From CH2014-Impacts (2014).

For rivers with a highly glaciated basin, mean summer runoff will not decrease as long as icemelt from the glaciers sustains the water flow, compensating for shorter and reduced snowmelt (see Figure 3.6; FOEN 2012). However, especially in small glaciated basins at lower altitudes, the effects of climate change on glacier runoff will be more pronounced (Huss and Fischer 2016). In such basins, summer flow will be strongly reduced over the 21st century, and stream intermittency will increase, because the contribution of glaciers is particularly important during low flow periods (Gurung and Stähli 2014, Stahl et al. 2016). In the very hot and dry summer 2003, the flow rates of basins with less than 10 % of glaciated area were below average (Bader et al. 2004). In August 2003, icemelt contributed to nearly 25 % of the Rhine discharge in Basel, and 75 % of the Rhône discharge in Geneva (Huss 2011). In future such summers, glacial runoff will significantly decrease as glacier storage progressively melts, affecting all the areas along the river course (Huss 2011). Other environmental variables will change in glacier-fed streams (see Figure 3.6). For example, turbidity will increase and conductivity decrease as glacial runoff first increases; later in the future, as a consequence of the reduction of glacial runoff, turbidity will decrease and conductivity will increase (Jacobsen et al. 2014).

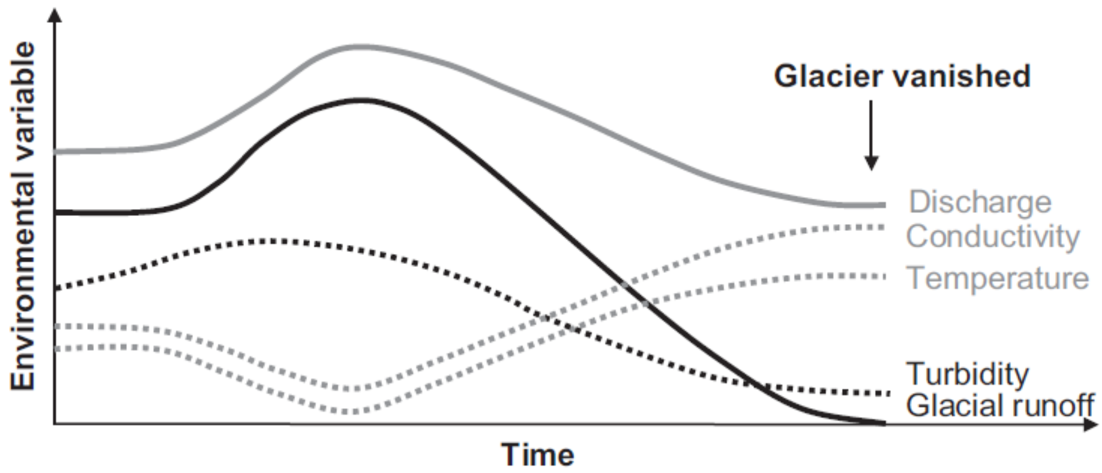


Figure 3.6: Conceptual diagram illustrating the expected temporal changes in selected environmental parameters at any point along a glacier-fed stream during glacial shrinkage. From Jacobsen et al. (2014).

Ice and snow cover of rivers will continue receding in the future (Magnuson et al. 2000, Klein et al. 2016), so that Alpine rivers will flow more continuously during the winter. For many such rivers, the future runoff will be more evenly distributed during the year, with an earlier peak of discharge (Etter et al. 2017), for example in March instead of April. In high alpine catchments, groundwater will continue to sustain the water flow in winter.

In summary, concerning low flows, minimal flowrates (e.g., measured by the Q347 flowrate, i.e., the flowrate that is exceeded 95 % of the year) are expected to decrease on the Swiss Plateau, but to increase (and occur earlier) in the Alps (FOEN 2012).

3.1.3 Altered erosion of banks and sediment transport

River bank erosion is directly correlated to a river's flowrate. Following changes in flowrate due to climate change (see above), erosion could increase in winter and decrease in summer for many rivers. This implies that the seasonality of transport of organic matter and sediments can also change accordingly (Murdoch et al. 2000).

In addition, glacier retreat and permafrost thawing liberates large amounts of sedimentary material. As a consequence, it is expected that the sediment load of rivers with high altitude catchments will increase in the near future, as runoff will progressively erode these formerly protected zones (Haeberli and Beniston 1998).

3.1.4 Other effects

In rivers with a pluvial to nival flow regimes and particularly in summer, climate change will likely increase the fraction of groundwater in the flow, due to the lesser contribution of rain and snowmelt (M. Schirmer, personal communication). This is likely to happen during summer droughts in catchments where there are well-buffered groundwater sources. This will affect the physicochemical properties of the water, specifically increasing the concentrations of major ions and the conductivity of the river water (Jacobsen et al. 2014).

Glacier recession affects stream lengths: in the formerly ice-covered areas, stream networks can lengthen (Finn et al. 2010, Robinson et al. 2014), such that novel and early successional river beds are created. These river sections typically lack riparian vegetation and are formed in the forefronts of the glacial melting zone. Robinson et al. (2014) observed river lengthening of 160 to 480 m between 1998 and 2009, and between 50 to 1300 m between 2009 and 2011, in several alpine catchments (e.g., Val Roseg).

3.2 Lakes

3.2.1 Increase of water temperature

Air temperature governs the radiative balance between the lake surface and the atmosphere, and is therefore a major determinant of water temperature in lakes (Peeters et al. 2002). Following air temperature increases, most lakes in the world are currently getting warmer (O'Reilly et al. 2015). Lake surface water temperature has been reported to increase at a mean worldwide rate of 0.34 °C/decade from 1985 to 2009 (O'Reilly et al. 2015). In Switzerland, lake water temperature is also increasing (Fink et al. 2014), with the mean temperature of Lake Constance increasing by 0.17°C/decade from 1962 to 1998, for example (Straile et al. 2003). In the deep water, consistent warming rates of 0.1–0.2 °C/decade have been observed in Europe (Dokulil et al. 2006). By the end of the 21st century, lake water is expected to be up to 4 °C warmer than today. The temperature increase will be most pronounced near the surface and in spring and early summer, when stratification is strongest. In lake Lugano, for example, the impact of global warming on the summer surface temperature during the 21st century is estimated to be +1–2 °C under a medium-emission scenario, or to +4–5 °C under a high-emission scenario (Lepori and Roberts 2015). There is, however, a lack of general data, and these warming trends are based on case studies rather than coordinated long-term monitoring across many lakes. In addition to the warming of individual lakes, the water temperature of lakes South of the Alps and North of the Alps has become more similar over the last century (Monchamp et al. 2017), which leads to homogenization of environmental conditions in lakes across the Alps (Pomati et al. 2017).

3.2.2 Altered evaporation

Lakes respond to higher surface temperature through an increase of the heat fluxes to the atmosphere, that is, increased longwave radiation and latent heat flux (evaporation) (Fink et al. 2014). Increased evaporation could be critical for very shallow lakes: their volume would be reduced and their residence time would increase (Schindler 2009), unless their inflows compensate the water loss. Most Swiss lakes do not face this issue, however small lakes or ponds (typically less than 1 km long and a few meters deep) will be more prone to drying during summer droughts (Brooks 2004).

3.2.3 Altered stratification, inflow and mixing regime, reduced freezing

In lakes, the surface waters (epilimnion) are more sensitive to air temperature than the deep layers (hypolimnion). Therefore, a consequence of higher air temperature is a higher temperature difference between the epilimnion and the hypolimnion during the warm summer season. In other words, with ongoing climate change, lake stratification tends to be stronger and to last longer (Beutel and Horne 2018). This may alter vertical mixing in low-altitude lakes: if the density of the water column does not adequately homogenize during winter (i.e., the epilimnion does not cool enough to match the hypolimnion's temperature), mixing cannot occur. Lake mixing, however, is especially important to provide oxygen to the deep water and to transport nutrients from the hypolimnion to the epilimnion (Posch et al. 2012, Yankova et al. 2017, Pomati et al. 2017), preventing nutrient accumulation in the deep water. Furthermore, regular and frequent vertical mixing prevents the build-up of anoxic, reducing conditions in the hypolimnion, which could, in turn, modify the interactions between water and sediments (Beutel and Horne 2018).

In low-altitude Swiss lakes, which are often monomictic (mixing once a year) or oligomictic (mixing every few years), vertical mixing is likely to occur less often than before. Higher altitude lakes, which are often dimictic (mixing twice a year) and often freeze over winter, are likely to switch towards monomictic regimes (Schindler 1997), and freezing will occur less frequently and less intensely, as reported over the last 150 years (Magnuson et al. 2000).

More frequent flooding in winter could bring more cold, oxygen-rich water to the deep water of lakes during this season (Fink et al. 2016, Råman Vinnå et al. 2018), which would counteract the effect of stronger stratification in lowland lakes. The future pattern of such deep intrusions depends on the characteristics of the lake-river system, and no general trend can be identified. Although not all factors were considered, the potential impact on oxygen renewal was judged to be negative for Lake Constance (Fink et al. 2016), positive for Lake Geneva, and uncertain for Lake Biel (Råman Vinnå et al. 2018).

Vertical mixing is also a function of wind: stronger winds result in a stronger exchange between surface and deep layers and can greatly favor the renewal of deep water in winter (Ambrosetti and Barbanti 1999). This was, for instance, observed in Lake Zug in December 1999 (FOEN 2016a). By fostering mixing, storms can also have important short-term impacts on the outflow and thus on downstream areas (Woolway et al. 2018). It is presently unclear how wind patterns will change in the future, making related processes difficult to predict.

Particularly in small lakes (< 5 km^2), the thermocline depth is also influenced by water clarity (Fee et al. 1996). This implies that thermocline development, sediment load, DOC concentration, biological activity, etc., are interlinked in such lakes, and that the change of one of these parameters can strongly affect the others.

During droughts, inflows decrease and the level of natural lakes decreases, thereby reducing outflows. In Switzerland, most large lakes are regulated, meaning that this effect can be dampened or avoided via thoughtful management, as was reported during summer 2003 (Bader et al. 2004).

3.2.4 Other effects

The absorption of solar light by lakes is strongly influenced by water clarity: in a turbid lake, solar light will be absorbed very quickly, whereas in a clear lake, it will reach greater depths. This has an important influence for warming of the surface layers and on light availability for aquatic organisms, with direct impacts on stratification and productivity (Schindler 1997). The first lakes downstream of alpine catchments are likely to receive more sediments, which are mobilized from melting glaciers and permafrost.

The formation of new lakes is often observed where glaciers recede, leaving a depression which fills with meltwater (e.g., Lake Trift, Lake Hüfi) (FOEN 2012). With the current and future strong retreat of glaciers in the Alps, the formation of many new lakes in high alpine catchments is to be expected (Haeberli et al. 2012).

3.3 Groundwater

This chapter primarily summarizes the major findings on groundwater from the National Research Programme NRP 61 (Hoffmann et al. 2014, Gurung and Stähli 2014). Two main groundwater types are considered, as they will react to climate change differently: aquifers receiving their water mostly from rain (precipitation-fed), and aquifers receiving their water mostly from rivers (river-fed).

3.3.1 Increase of water temperature

Even though groundwater is not directly in contact with the atmosphere, its temperature will increase with climate change (Taylor et al. 2013). This will be especially pronounced for river-fed groundwater, which could warm by up to 3.5 °C by the end of the 21st century (CH2014-Impacts 2014). Precipitation-fed groundwater is more buffered and is expected to warm by up to 1 °C in the same period (CH2014-Impacts 2014). In karst aquifers in the Jura mountain range, a temperature increase of ~0.5 °C between 1989 and 2012 was reported (MFR and ISSKA 2012). Other factors such as altitude, exposure, land use, aquifer type and depth, and anthropic impacts influence the future trend of groundwater temperature (Schürch et al. 2018).

3.3.2 Altered recharge regime

Climate change may modify the water balance (through altered precipitation and runoff regimes; see Sections 2.2 and 3.1.2), therefore affecting groundwater resources. Aquifers react very differently to dry/wet periods, depending on their water sources and geological characteristics; some general considerations are given here.

Currently, recharge of precipitation-fed groundwater occurs mainly in fall, winter, and spring. In summer, evapotranspiration is high so that only negligible amounts of water reach the groundwater, except during strong storms. Drier and warmer summers will therefore not strongly influence the recharge rate of precipitation-fed groundwater. However, water losses may increase (Kipfer and Livingstone 2008), notably due to higher evapotranspiration. This will likely extend the period without recharge, although this may be counteracted by the predicted increasing strength of summer storms. Meanwhile, river-fed groundwater will respond to variation in river flow, and thus recharge is expected to decrease in late summer but increase in winter and early spring (Eckhardt and Ulbrich 2003), depending on how the flowrate regime of the source rivers change in the future.

Groundwater recharge was reported to be more affected by extreme events than by changes in mean precipitation volume (Taylor et al. 2013). If the frequency of extreme precipitation events increases in Switzerland, recharge could locally increase, especially over winter and for precipitation-fed groundwater. However, no global trend can be identified, as such events are difficult to predict (Scherrer et al. 2016). In catchments where surface runoff is a primary pathway (e.g., alpine areas), extreme rain does not contribute much to recharge rates. Alpine aquifers, for which snow cover prevents or reduces recharge in winter, are likely to receive more water during future winter conditions, but less during the late spring (due to less snowmelt, and possibly reduced precipitation). Enhanced recharge can also be expected in permafrost areas due to thawing (Dragoni and Sukhija 2008).

4 Land use and water quality: Context in Switzerland

Land use largely determines the chemical inputs to waterbodies. Exports of nutrients (nitrogen, phosphorus, potassium) and other anthropogenic chemicals from intensively-used agricultural land and from urban areas are significantly higher than from other types of land uses (Zobrist and Reichert 2006). Small lowland streams are particularly exposed to inputs of these compounds, as well as of micropollutants (Strahm et al. 2013). Human impact on water quality in forested and alpine catchments is generally very low and it is unlikely that climate change will fundamentally affect this; however, climate-induced alterations may still be ecologically relevant, despite being small in absolute terms. Below we briefly summarize the relevant characteristics of the main catchment types (urban, agricultural, forested and alpine). In Section 5, we discuss the processes specific to each of these catchment types.

Urban areas are hotspots of waterbody pollution, both through point sources (wastewater treatment plants, industrial discharges, etc.) and diffuse sources (untreated street runoff, waste disposal, etc.). Thanks to better wastewater management and a series of regulations, pollution from urban areas has been greatly reduced over the last decades. However, the number of chemicals has been steadily increasing (M. Schärer, personal communication), and this variety makes both removal and toxicity assessment difficult.

Agricultural lands typically receive significant amounts of fertilizers (rich in nitrogen (Lin et al. 2001) and phosphorus) and plant protection products; high concentrations of specific chemicals in streams and lakes can sometimes be directly related to nearby agricultural activities (Royer et al. 2006, Zobrist and Reichert 2006). In Switzerland, waterbody pollution by adjacent agricultural fields is a primary water quality issue (M. Schärer, personal communication). Small streams can contain high pesticide concentrations, above ecotoxicological thresholds (Doppler et al. 2017). In addition, some middle-sized lakes such as Lake Zug (FOEN 2016a), Lake Baldegg, and Lake Sempach (Müller et al. 2012) still have not fully recovered from the eutrophication of the 1950s to the 1970s because of the high nutrient inputs (in particular phosphorus) from their mainly agricultural basins, despite reduction measures during the last decades. Nutrient inputs can deteriorate water quality in stratified lakes by favoring algal growth at the surface and oxygen depletion in the deep water (Foley et al. 2012).

Forests in Switzerland consist of deciduous trees (e.g., the common beech) in the lowlands, while mountainous areas at altitudes from about 800 to 2000 m are largely dominated by coniferous trees (in particular the silver fir and the Norway spruce). Overall, tree cover in

Switzerland has increased by about 10 % between 1985 and 2013 (stable on the North Plateau and Jura, 20 % on the South side of the Alps and up to 28 % in alpine regions) (Abegg et al. 2014). Forested catchments are characterized by high evapotranspiration (which is particularly high in summer and sustained during dry seasons) and therefore less water loss through runoff and soil infiltration. Forests can be seen as a buffer for most chemicals (which are retained by the soil and the trees) and therefore as a valuable filter for our water resources (F. Hagedorn, personal communication).

Alpine areas are significant in Switzerland: nearly one quarter of the surface of the country lies above 2000 m. Here, we focus on natural alpine areas, i.e., excluding mountain pastures. Alpine catchments are characterized by a large fraction of rocky, permafrost, and ice-covered areas, with snow cover often lasting most of the year. Except for scree areas, high alpine surfaces often have a relatively low permeability, which, combined with the steep slopes, results in an amplified runoff response to rain. Given the low anthropogenic activity, alpine catchments are rather indirectly affected through environmental processes. Sediment exports from alpine catchments are a significant source for downstream areas, and climate-induced changes could have consequences in the lowlands as well.

Nearly 40 % of the total Swiss river network length is influenced by urbanized areas, at least 65 % by cultivated fields, and 90 % by grasslands, including pasture (Strahm et al. 2013).

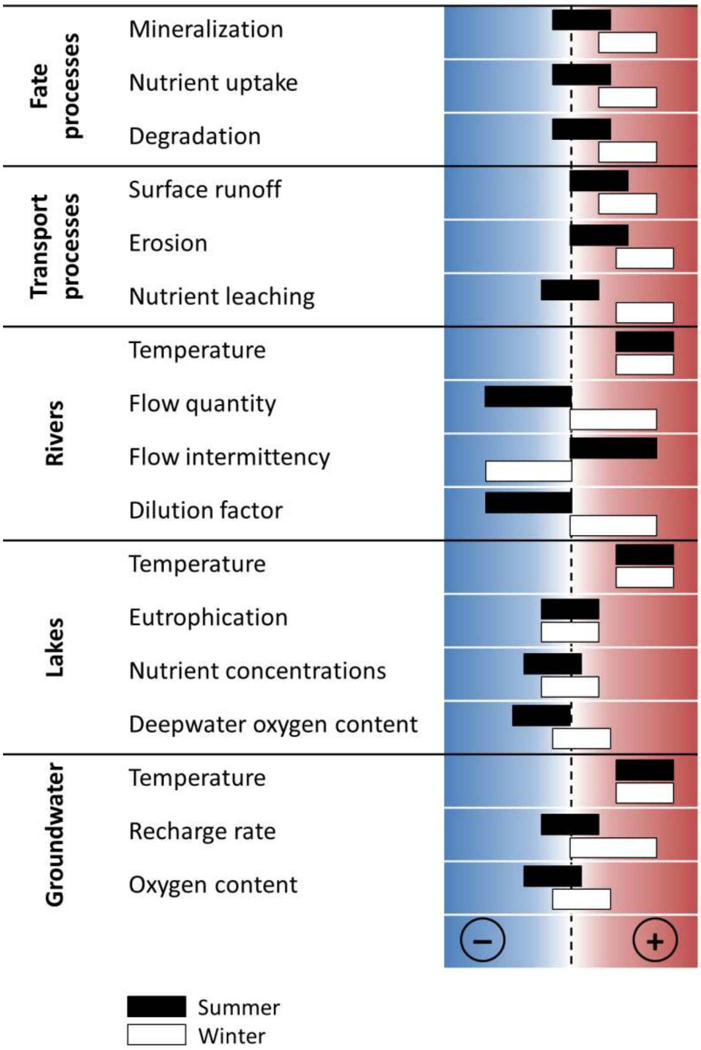


Figure 4.1: Qualitative impact of climate change on quantities and processes related to water quality.

5 Impacts of climate change on catchment processes

Box 3: Synthesis – Impacts of climate change on catchment processes relevant for water quality.

Swiss catchments will be increasingly impacted by climate change. Above all, severe droughts and heavy rainfall will modify the hydrological, thermal and chemical regimes, which will affect waterbodies.

The runoff regime will change considerably in Swiss catchments, which are mostly controlled by rain and snowmelt: increased flow in winter, reduced snowmelt, and generally reduced flow in summer and autumn. The latter will be amplified by increased evapotranspiration. As a result, chemical exports (runoff and leaching, e.g., of nitrate) will increase in winter and decrease in spring and summer. The decrease in spring will be particularly marked in high alpine catchments, where snowmelt peaks can represent a major nutrient source.

Waterbodies will experience more extreme conditions (e.g., high and low flow) and receive stronger pulses of pollutants – in particular, sediments in alpine rivers and anthropogenic pollution in urban and agricultural catchments. In groundwater, severe droughts will be followed by increased loading of weakly-sorptive compounds (typically, nitrate), which accumulate in soils and are easily mobilized by subsequent leaching – such increased loading can sometimes be observed for several months. The effect will be most pronounced for aquifers in agricultural catchments. In addition, higher temperatures will increase evapotranspiration throughout the year, extending the period without groundwater recharge.

Despite these predicted changes, climate-induced changes are not expected to result in catchment alterations that strongly impair water quality for direct human use in Switzerland beyond the magnitude of other anthropogenic impacts during the next decades. However, there may be significant alterations (e.g., sediment regime in alpine lakes and rivers, or community changes in forests), which could lead to important but hardly predictable ecological changes.

Waterbodies are strongly influenced by the processes in their catchment. For instance, the relative importance of surface runoff versus infiltration – which is temporally and spatially variable – affects the timing and chemistry of the water reaching aquifers, rivers, and lakes. The hydrological and chemical behavior of catchments will be impacted by climate change, with consequences for fate and transport processes (Murdoch et al. 2000). These impacts are qualitatively summarized in Figure 4.1.

5.1 Fate processes

Environmental conditions affect biochemical processes and equilibria. In this section, we summarize how climate-induced changes affect these processes, including chemical kinetics, soil conditions and processes, organic matter and nutrient cycling, and transformation of chemicals. The fate of chemical compounds in the soil can often be linked to their transport to waterbodies, which will be discussed in the following section.

Virtually all biochemical reactions are influenced by temperature, water availability, and oxygen. Soil temperature and soil moisture thus largely determine the fate of chemicals. In addition, specific processes such as droughts and freeze-thaw cycles are stressful for plants and soil microorganisms (Schimel et al. 2007), and thus also play an important role. Topsoil is a particularly relevant setting for fate processes, as it hosts intense microbial activity when conditions are favorable (during the growing season); the bulk of annual carbon and nutrient cycling occurs in the topsoil (Kalbitz et al. 2000).

As **temperature** increases by 2 to 4 °C, many reaction rates increase, typically by 10 to 40 % (following the Arrhenius relation, see, e.g., Davidson and Janssens (2006)). Different reactions accelerate at different rates, so that a temperature increase can favor some reactions over others. In addition, biological processes are limited by the optimal range of organisms: their rates will decrease beyond a certain optimum temperature. Generally, most metabolic processes will thus proceed faster under a warmer climate, resulting in faster production, assimilation, transformation and degradation of chemicals. As a result of accelerated biochemical reactions, gas release from soils is also temperature-sensitive and increases at higher temperatures (Johnston et al. 2004).

Soil moisture, like temperature, generally correlates positively to reaction rates. However, high soil moisture is linked to low oxygen content, which hinders aerobic reactions and thus slows down the oxidative transformation of compounds. In summer, warmer temperatures and increased drought severity will result in a general reduction of soil moisture. Under such conditions, many microbial processes in aerobic soils will be slowed down as soil moisture decreases (Schlesinger et al. 2015). In summer, this counteracts the effect of increased temperature, although higher process rates may then occur under reestablished moisture conditions. In anaerobic soils, reduced water levels may allow oxygen presence and thereby halt anaerobic microbial processes (e.g., denitrification) but favor aerobic ones (Schlesinger et al. 2015). In winter, under the assumption that conditions become wetter (more precipitation, less snow and freezing), biochemical processes will be favored, intensifying the effect of increased temperature.

Organic matter cycling will be affected by these drivers. Under warmer temperatures and higher atmospheric concentrations of CO_2, most plants will grow faster, while soil bacteria will also benefit from warmer conditions. As a consequence, there will be an increase in the rate of organic matter production and decomposition (Kalbitz et al. 2000), resulting in increased release of DOC into the soil (Evans et al. 2005, Bardgett et al. 2008, Hagedorn et al. 2018) and increased export of CO_2 to the atmosphere (Kirschbaum 1995, Davidson and Janssens 2006). During future summer droughts, these processes are likely to be inhibited despite the warmer conditions, as shown in field experiments (Hagedorn et al. 2018). The net effect on soil C reserves is difficult to assess globally, and is related to many specific properties of the soil, vegetation, and climate. However, both observational and modeling studies point towards increased C losses in the future (in particular in non-fertilized, non-forested soils), and therefore reduction of the soil C reserves (Hagedorn et al. 2018). A reserve diminution of ~14 % between the 1980s and 2011 (equivalent to 0.4–0.9 tons C/ha/year) was reported in forested alpine soils in Germany, and this loss is similar in Swiss forests (Hagedorn et al. 2018). The impact on waterbodies will likely remain insignificant.

Overall **nutrient** uptake by plants is likely to get more efficient in the future climate (Boxall et al. 2009, Bernal et al. 2012). It is also expected that warmer-wetter conditions will stimulate bacterial activity, and thus mineralization of organic N (Arheimer et al. 2005, Gurung and Stähli 2014) and organic P (Kalbitz et al. 2000), nitrification, and denitrification (Bernal et al. 2012). In parallel, higher temperatures could increase N losses to the atmosphere as N_2O (Bardgett et al. 2008, Hagedorn et al. 2018). Overall, the N cycle will accelerate, except during very dry periods. Indeed, if moisture declines in aerobic soils, microbial oxidative processes (e.g., nitrification and N_2O production, litter decomposition) slow down (Schlesinger et al. 2015). Tree roots become deeper, so that nutrients, base cations, and trace elements are also taken up from deeper soil layers (Schlesinger et al. 2015). As a result, an overall decrease in availability of carbon, nutrients (e.g., N, P) and base cations (e.g., Ca^{2+}, Mg^{2+}, K^{+}) can be expected during droughts (Schlesinger et al. 2015).

Freeze-thaw cycles are known to impact carbon and nutrient dynamics in soils (Matzner and Borken 2008). For N and P, there is agreement that the occurrence and severity of freeze-thaw cycles correlate with increased exports (Fitzhugh et al. 2001, Callesen et al. 2007). Soil freezing was often shown to enhance nitrate release upon thawing (Callesen et al. 2007), likely because of increased fine root and microbial mortality (Edwards et al. 2007), reduced plant N uptake, and reduced competition for inorganic N (Groffman et al. 2001). Soil freezing disrupts the coupling between nutrient production (mineralization, and nitrification for nitrate) and consumption (plant uptake) by strongly reducing the latter, making more nutrients available for transport.

Xenobiotics (i.e., artificial chemicals) will also react to changes in soil conditions. In the pore water of the soil, increased temperature will favor hydrolysis of the chemicals sensitive to this degradation process (Noyes et al. 2009). Hydrolysis results in transformation products that can be more or less active (and undesirable) than the initial compounds (Noyes et al. 2009). It can be expected that degradation rates will increase particularly strongly for chemicals for which the contact with the soil is long and intense (e.g., transported by matrix flow rather than by preferential flow – see Section 5.2; Boxall et al. 2009). In drier soils, a reduction of hydrolysis is observed (Schlesinger et al. 2015), again opposing the effect of increased temperature during droughts. Mineral weathering is also reduced under drier conditions (Schlesinger et al. 2015), resulting in a lower availability of soil and surface minerals for transport.

Within the soil matrix, many chemicals are sorbed (primarily to organic matter), thus being temporarily removed from the hydrological cycle. As mentioned above, soil OC may decrease in the future, which would lead to decreased retention of sorptive chemicals (e.g., pesticides) (Boxall et al. 2009). The net effect on storage and availability of sorptive chemicals will probably be weak.

To summarize, temperature increase will accelerate fate processes, stimulating nutrient cycling and biochemical degradation. Reductions in soil moisture (typically during drier summers) will counteract this acceleration. The overall effect will vary between regions and seasons, as a function of the magnitude and seasonality of changes of these two drivers. In the end, nutrient availability will evolve depending on the balance between microbial mineralization (which makes nutrients available for the hydrological cycle) and plant uptake (which removes them). These changes will be gradual and may not necessarily be significant over the next decades (V. Prasuhn and W. Richner, personal communication).

5.2 Transport processes

There are several processes by which water and chemicals are transported through catchments. The most important ones are runoff (surface flow of water), erosion (solid particles entrained by water or wind), and infiltration into preferential flowpaths and into the soil matrix (causing leaching of chemicals). These different transport processes are characterized by different chemical signatures. In this section, we describe the possible impacts of climate change on these transport processes, focusing on their relevance for water quality in waterbodies.

Rainfall influences the transport of chemicals such as nutrients and pesticides to waterbodies (Bloomfield et al. 2006, Boxall et al. 2009, Hoffmann et al. 2014). A shift in rainfall quantity or partitioning into the different transport processes can affect water quality. Increases in rainfall usually bring more contaminants directly into lakes and rivers, via both runoff and eroded particles (Schindler 1997, Bloomfield et al. 2006, Jeppesen et al. 2009, Noyes et al. 2009). In addition, larger rainfall events amplify water infiltration through the soil, which transports chemicals through preferential flowpaths (macropore flow) or more slowly through the soil matrix (matrix flow). Summer droughts, when high temperatures and dry conditions prevail, strongly affect transport processes. During droughts, evapotranspiration increases, and water flows are reduced, resulting in reduced export of chemicals to waterbodies and increased concentrations of compounds (e.g., DOC) in the topsoil (Kalbitz et al. 2000). As a result, rainfall after a prolonged drought period is thus often described as a critical situation (V. Prasuhn and W. Richner, personal communication), as the chemicals accumulated during the drought are rapidly remobilized and flow into waterbodies (Murdoch et al. 2000, Skjelkvåle et al. 2003, Evans et al. 2005), reaching peak loadings and possibly extreme concentrations. Such chemical pulses (e.g., DOM or nitrate) are also observed after soil thawing or snowmelt, when frozen material is released into the hydrological cycle (Kalbitz et al. 2000, Skjelkvåle et al. 2003). For groundwater, rising water levels after long dry periods can also induce concentration pulses as the rising groundwater washes away chemicals from soil layers that were previously above the water table (Gurung and Stähli 2014).

Macropore flow, surface runoff and erosion are the most important transport pathways for particulates and sorptive compounds (e.g., phosphate, ammonium, hydrophobic chemicals, i.e., most pesticides, bacteria and viruses, heavy metals) (Boxall et al. 2009). These transport pathways are active during heavy rains and (rapid) snowmelt, and after droughts (Kalbitz et al. 2000, Royer et al. 2006, Boxall et al. 2009). In the latter case, soils may have become more hydrophobic, thereby reducing infiltration and enhancing these processes during the next rain. These transport pathways will likely gain importance during future, more severe storms. Regarding soil erosion, several climatic drivers will contribute to this process: (i) in the lowlands, higher rain quantities in winter and more frequent heavy rains, (ii) in the highlands, more frequent heavy rains in summer on a thawing, less stable soil with shorter snow cover (Scheurer et al. 2009). Increases in erosion lead to higher P transport, which is often adsorbed to soil particles (Sharpley et al. 2001). Additionally, the occurrence of mass movement (landslides, mudflows, rockfall, etc.) will also increase with more extreme precipitation patterns. Both soil erosion and mass movement are linked to sediment loading in rivers and lakes, with high ecological relevance (Scheurer et al. 2009). Alpine and agricultural areas are particularly exposed to erosion.

In lakes, which act as a buffer between the headwaters and the downstream areas, more runoff generally results in higher chemical concentrations. For example, P loading, a trigger for lake eutrophication (Fitzhugh et al. 2001), is expected to increase where rainfall quantity and/or intensity increases (Jeppesen et al. 2009, Carpenter et al. 2017). Nitrate runoff, which can cause surface water acidification (Campbell et al. 2000, Skjelkvåle et al. 2003), also increases following intense rainfall or snowmelt (see Section 5.6); this has also been reported in Swiss alpine waterbodies (Steingruber and Colombo 2006). Drier conditions in summer would have the opposite effect: concentrations of chemicals in runoff from diffuse sources would be reduced in many catchments, however, these concentrations may increase temporarily at the onset of a wet period. During drier summers, lakes would therefore receive globally less chemical inputs.

Matrix flow is often the most important transport pathway for dissolved chemicals (e.g., DOC, nitrate, major ions, weakly-sorptive pesticides) (Boxall et al. 2009). It was reported that increased rainfall and more frequent heavy rain were responsible for higher ion diffusion rates (Schlesinger et al. 2015), increased leaching of nutrients to the groundwater (Gurung and Stähli 2014) and increased remobilization of persistent chemicals such as POPs (Kallenborn et al. 2012) from soils. Weakly-sorptive chemicals – typically nitrate (Lin et al. 2001) – can be readily transported to the groundwater. Leaching is particularly important if the soil is wet and the throughflow is large (e.g., under rainy conditions). However, dry conditions may put the vegetation under water stress and reduce their nutrient uptake, which may then also lead to increased subsequent nutrient leaching (Prasuhn and Albisser 2014).

As discussed in the previous section, higher temperatures would enhance nutrient availability in soils, in particular if soil moisture remains sufficient. As a result, increased leaching of different elements can be expected, possibly exacerbating the effect of the wetter conditions predicted in future winters. Increased microbial decomposition of organic matter and increased plant growth could lead to greater leaching of DOC to the groundwater (V. Prasuhn and W. Richner, personal communication; Bardgett et al. 2008). In future winters, increased mineralization of organic N (see Section 5.1) would result in higher root-zone N concentrations and possibly in increased leaching of nitrate (Murdoch et al. 2000, Arheimer et al. 2005). This effect, however, could be reduced or even reversed by faster N uptake by vegetation (Bernal et al. 2012, Prasuhn and Albisser 2014).

Decreased soil moisture, expected in future summers, would reduce leaching. During droughts, although fate processes are also slower, chemicals are observed to accumulate in soils. When the first rains rewet the soil, and for several months subsequently, higher chemical concentrations are observed in export fluxes (Kalbitz et al. 2000, Whitehead et al. 2009, Schlesinger et al. 2015). Thus, by increasing drought severity in summer, climate change could result in higher chemical fluxes towards waterbodies in the following seasons (fall and winter). This impact would be amplified by wetter winters. Weakly-sorptive compounds such as nitrate are released rather rapidly, while strongly-sorptive compounds such as phosphate can remain bound into the soil matrix over significantly longer times.

As leaching is favored if soil moisture is high, its seasonality may change, generally becoming higher in winter and lower in summer (Bloomfield et al. 2006). In monitored rivers in Switzerland, nitrate concentrations were below average during the summer drought of 2003, but they were above average – although not exceptional – over the following winter (Figure 5.1 shows the example of the Rhine).

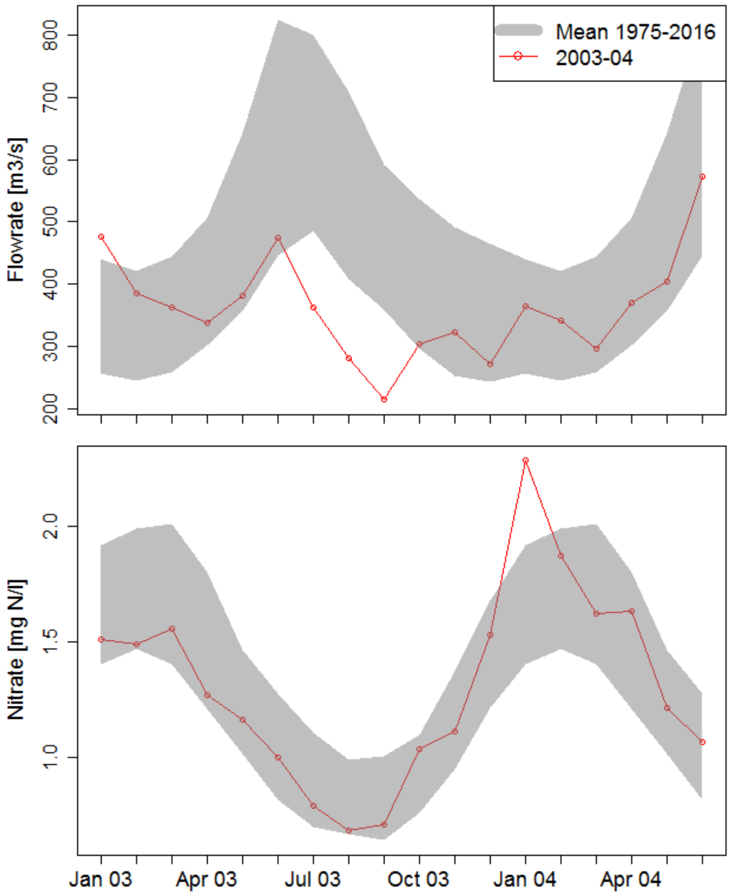


Figure 5.1: Monthly values of flowrate (top) and nitrate (bottom) in the Rhine at Rekingen, Switzerland. The grey area shows one standard deviation above and below the monthly means over the whole dataset (1975–2016). The red line shows the monthly means in 2003–04, with summer drought conditions in 2003. Data source: NADUF monitoring program (biweekly data).

More recharge of precipitation- and river-fed groundwater in winter would result in higher water levels in the spring; however, drier summers would reduce these levels during the second half of the year. Longer droughts are expected to impact groundwater chemistry. During dry periods, fewer chemicals move through the soil, and short-lived contaminants are degraded before reaching the aquifer (Hoffmann et al. 2014). The first major rainfalls may however feed the groundwater with significant amounts of DOC, nutrients, and contaminants that accumulated in the soil during the drought (Evans et al. 2005, Hoffmann et al. 2014).

As discussed in the previous section, freeze-thaw cycles affect nutrient availability. In the lowlands, milder winters with less soil freezing would allow more plant activity and could therefore reduce nutrient leaching, but slightly increase C losses during winter (Matzner and Borken 2008). At moderate elevations (from ~1000 to 2500 m), reduced snow cover from autumn through spring will lead to more freeze-thaw cycles, potentially increasing nutrient leaching (Fitzhugh et al. 2001, Matzner and Borken 2008). The effect at higher elevations depends greatly on the local patterns of future snow cover (Edwards et al. 2007).

To summarize, rapid transport processes (macropore flow, runoff, erosion and mass movement) are likely to gain strength during future, stronger summer storms. These storms will occur in the context of (i) more severe droughts, with more chemicals available for transport, (ii) warmer soils (possibly thawing), with more sedimentary material available for transport. Overall, recurrent heavy rain and more severe droughts will make chemical pulses more frequent and stronger over summer. Many rivers may periodically experience higher loads of sediments, phosphate, pesticides, wastewater, or other human contaminants. Slow transport processes (matrix flow) are likely to be enhanced during wetter winters (Boxall et al. 2009), but reduced during drier summers. This will increase ion transport and nutrient leaching in winter (e.g., nitrate), which may be partly counteracted by fewer freeze-thaw cycles at low elevations, but favored by more freeze-thaw cycles at moderate elevations with reduced snow cover. Figure 5.2 illustrates these relationships.

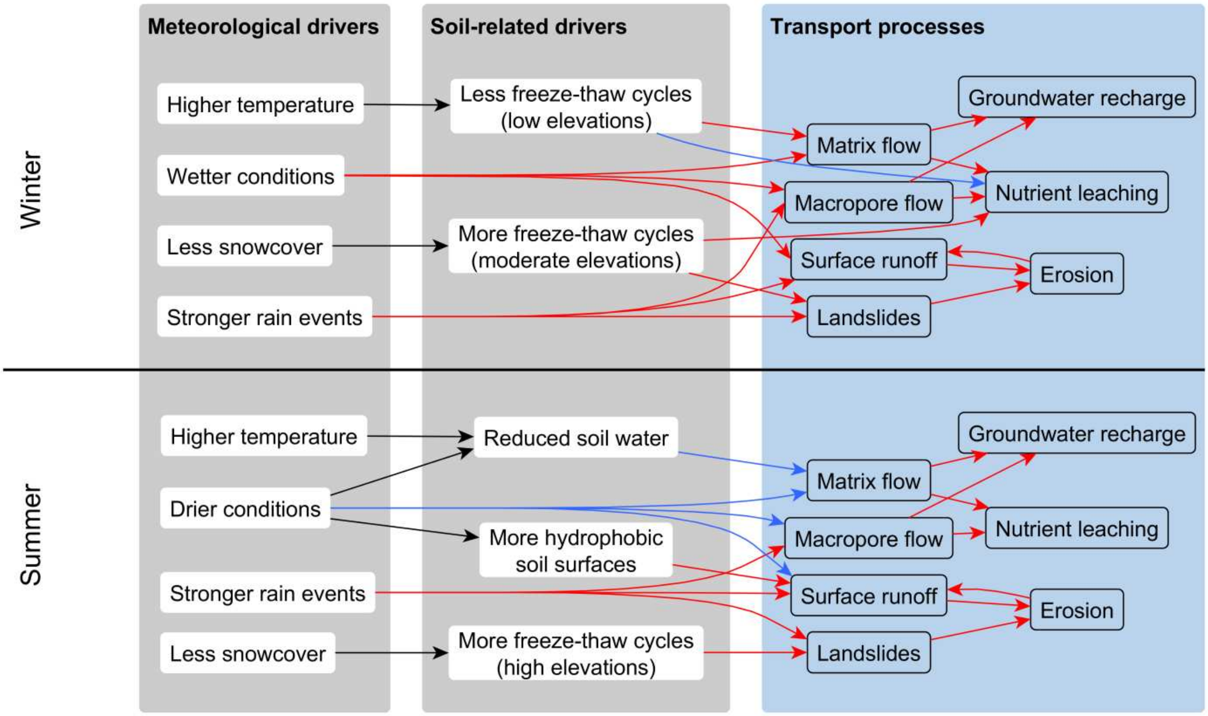


Figure 5.2: Conceptual diagram linking the drivers (left and middle panes) to their positive (red arrows) or negative (blue arrows) impacts on transport processes (right pane).

5.3 Alterations in urban areas

Urbanized areas are well known for being warmer than the surrounding environment, due to (i) dark surfaces (asphalt, concrete, etc.) which absorb solar radiation, (ii) low evapotranspiration, and (iii) human activities releasing heat. Generally speaking, runoff originating in urban areas will therefore be warmer than it would be in a more natural environment (Rossi and Hari 2009). A typical example are summer storms, where large amounts of rain fall on warm surfaces, creating warm runoff which may flow into sensitive waterbodies such as small streams (Herb et al. 2008). In the future, such thermal pollution will occur under even higher temperatures and thus become more severe for waterbodies. Increased use of waterbodies for space and process cooling could further exacerbate this effect (Gaudard et al. 2018). Given the high urban density, Switzerland will be increasingly exposed to these issues. Three case studies in Switzerland showed that runoff from roofs or streets can abruptly increase stream temperature by up to 4 °C, however temperatures rapidly decrease after onset of the storm (Rossi and Hari 2009). This was confirmed by observations in the US, which found brief temperature increases > 7 °C (Nelson and Palmer 2007).

Storms can also be a primary source of contaminants for the nearby waterbodies (VSA 2007), such as PCBs (Rossi et al. 2004), waste, polluted sediments (Rossi et al. 2013), surface pesticides, and heavy metals (Barnes et al. 2002). Three case studies in Switzerland showed significant increases in river sediment concentrations of metals and polyaromatic hydrocarbons downstream of stormwater discharges (Rossi et al. 2013). In addition, longer droughts will favor accumulation of pollutants on urban surfaces, which accentuates pollution load during the first rain. It is worth noting that such pollution events are particularly critical in the case of small rain events, which wash out the pollutants but do not sufficiently increase the dilution capacity of the recipient waterbodies (M. Maurer and A. Joss, personal communication). A model for Western Switzerland predicted higher pollutant concentrations in urban runoff in future summers and lower concentrations in future winters, due to changes in the dilution ratio (Coutu et al. 2013; see Section 6.1.4).

In many urban areas, street runoff is mixed with wastewater (so-called combined sewer systems), so that during strong rain, total runoff may exceed the capacity of the wastewater treatment plants. This situation results in untreated wastewater, together with street runoff, being discharged directly to the environment (combined sewer outflows or CSOs). These events can bring considerable amounts of nutrients and contaminants into surface waters and temporarily worsen water quality (Heinz et al. 2009). A study in Greifensee in northern Switzerland revealed that, annually, 2 to 3 % of the wastewater is discharged untreated in lake tributaries because of CSOs (Buerge et al. 2006). For compounds such as phosphorus that are efficiently eliminated in wastewater treatment plants, this input pathway to the lake is of similar importance as treated wastewater (Buerge et al. 2006). This issue is likely to become more pronounced in regions where precipitation is expected to intensify. In Switzerland, intense storms (at any time of the year) are likely to push water treatment infrastructure to its limits more and more often, thereby causing more pollution peaks in waterbodies (P. Niederhauser and M. Schärer, personal communication). This fact that climate change will increase the severity of CSOs is widely recognized, however the quantitative impact is likely to be regional. In a case study in Norway, CSOs were simulated to increase 1.5 to 3 times faster than precipitation increased (Nie et al. 2009). However, a study in Quebec found that the number of CSO events should remain constant by the end of the 21st century, but that their duration would increase slightly (Fortier and Mailhot 2015). In the Great Lakes region, increasing precipitation was modeled to increase the frequency of CSOs into Lake Michigan by 50–120 % by the end of the 21st century, possibly impacting drinking water supplies and recreational beaches (Patz et al. 2008).

Finally, the occurrence of exceptional rain (i.e., events with a large return period, exceeding the capacity of the hydrological network) may become more frequent (Milly et al. 2002). This will tend to increase the risk of exceptional pollution from inadapted infrastructure, due to the damage caused by such extreme events.

5.4 Alterations in agricultural lands

An increase in severity of heavy rain will also impact agricultural land, with increased erosion (Fuhrer et al. 2006, IPCC 2007), extreme runoff, and drainage, all of which can considerably increase contamination of waterbodies from fields (Sharpley et al. 2001, Donald et al. 2007). Indeed, it was shown that substantial P losses to waterbodies can originate from agricultural areas during large storms, when high soil P (due to fertilizer application) coincides with elevated surface runoff, subsurface flow or drainage, and erosion (V. Prasuhn and W. Richner, personal communication; Sharpley et al. 2001). The predicted increase in occurrence of summer storms is likely to make such losses more significant, for P and for most pesticides.

Leaching is generally a minor transport pathway for pesticides (Flury 1996). For nitrate, increased mineralization in winter could lead to increased leaching during winter and spring (Gurung and Stähli 2014), which could result in an increase in N export to waterbodies from agricultural lands (Arheimer et al. 2005).

Rainfall induces transport of plant protection products (herbicides, fungicides, insecticides) from agricultural lands to the nearby waterbodies. The amount of such products washed away depends on how soon after application the first rain occurs, and how strong the rain is. Insecticides, which are particularly critical regarding water quality, are usually applied during insect outbreaks, which could become more frequent in a warmer climate. Exports of insecticides and their degradation products may increase after dry periods, as they accumulated over a long time.

Under higher temperatures, photodegradation (in water) and biodegradation (in water and soils) of pesticides will proceed faster (Bloomfield et al. 2006, Noyes et al. 2009, Huntscha et al. 2018). Degradation produces other compounds (metabolites), which are usually less toxic than the parent compounds but more mobile and persistent (Bloomfield et al. 2006). Production reactions will also be reinforced by warmer weather: for example, increased production of mycotoxins (toxic metabolites; Boxall et al. 2009), typically from crop-colonizing fungi, can occur.

In karst aquifers, strong rain (especially in summer) is often linked to microbial contamination and degradation of water quality at the springs (Besmer et al. 2017), as the water flows rapidly through the karst network. Such degradation is primarily an issue for karst systems within agricultural catchments (U. Von Gunten, personal communication). In Switzerland, karst spring water is treated when used as domestic water, and an increased frequency of these events is not expected to affect water quality in the long term.

To summarize, two main climate-induced processes are likely to increase the exports of nutrients and pollutants from agricultural lands to waterbodies: (i) faster N mineralization, enhancing availability of nitrate for transport, (ii) heavy rains causing rapid infiltration, runoff and erosion, and thus rapid chemical losses. Several other mechanisms may balance these exports: (i) faster nutrient uptake by plants, reducing availability for leaching, (ii) faster degradation of chemicals, less of which reach waterbodies, and (iii) more frequent droughts, during which transport is reduced – although it is then often enhanced during soil rewetting. Considering climate change alone, chemical transport can be expected to decrease in summer, and increase in winter and during storms (including summer storms). However, changes will strongly depend on other, more direct human impacts, so that the overall trajectory is difficult to predict (see Section 15.1).

5.5 Alterations in forested catchments

With climate warming, coniferous stands move progressively higher in elevation: they will be increasingly pushed away from the lowlands by better-adapted species, but will profit from better conditions in alpine regions. Given the long lifetime of trees, community changes may take several centuries. It is worth noting that DOC and DON exports from coniferous forests were reported to be significantly higher than for deciduous ones (Kalbitz et al. 2000).

Droughts have complex effects on processes in forest ecosystems, which should be considered given the foreseen increase in drought frequency in many regions. During droughts, many tree species reduce nutrient uptake and C sequestration, which can lead to increased C losses (Schlesinger et al. 2015).

In addition, droughts will result in higher occurrence of forest fires in some regions, which increases soil erosion (Lindner et al. 2010), loss of organic material (Hagedorn et al. 2018), export of N (Schlesinger et al. 2015), P, and possibly heavy metals to waterbodies, and increases the exposure of small lakes to wind (Schindler 2009). However, the fire-related leaching of nutrients to waterbodies is small in comparison to atmospheric losses (Schlesinger et al. 2015). In Switzerland, forests are rather scattered, limiting the risk of large-scale destruction. Locally however, more frequent fires are to be expected in drier and hotter summers. More frequent and stronger storms will also increase average forest destruction through windthrow (Fuhrer et al. 2006), especially in wind-exposed areas. Another threat to forests is insect pests such as beetles, which are expected to survive better through warmer winters, potentially increasing forest destruction (Schindler 2009).

Forest destruction in general (by fire, storms, or beetles, and also by other human activities and by changes in local conditions) exposes soils to warmer temperatures, enhances degradation of labile humus, and increases the amount of organic material available for leaching, runoff and erosion (Schlesinger et al. 2015). Reduced forest area or density means less evapotranspiration and chemical cycling, and thereby increased exports of water in the short term, but, in the long term, decreased exports of nutrients (Bernal et al. 2012). In Switzerland, it is likely that climate-induced forest destruction will be of minor significance compared to land use management, and that colonization (e.g., at higher elevations) and natural reforestation (e.g., of former alpine pastures) will continue to offset deforestation.

To summarize, climate change will have progressive, strong impacts on forest communities. Replacement of coniferous forests by deciduous ones could lead to a reduction in DOC and DON exports, while increased forest destruction could result in higher inputs of organic matter to downstream waterbodies over short time scales, but lower inputs in the long term.

5.6 Alterations in alpine catchments

Glaciers throughout the world are melting due to climate warming. In glaciated catchments, increased glacier and permafrost melting could result in a cooler summer runoff (Rouse et al. 1997, Wang et al. 1999). This effect would only be temporary, and last only as long as enough ice remains; in Switzerland, such cooling may last until the end of the 21st century in the watersheds with the larger glaciers. As permanent snowpack or ice melts (in particular in summer), there is also a release of the contaminants (e.g., POPs) stored there after atmospheric deposition (Noyes et al. 2009, Kallenborn et al. 2012).

A second impact of ice and permafrost melting is the exposure of many high altitude areas to erosion, especially from mid-summer to early autumn. It is thus expected that the amount of sediments reaching waterbodies and downstream areas will increase in the future (Rouse et al. 1997, Wang et al. 1999, Gurung and Stähli 2014), which will be amplified by longer summer flooding periods in alpine regions (Köplin et al. 2014). This has been clearly observed in the Rhône catchment in Switzerland as a response to the temperature shift at the end of the 1980s (Figure 5.3; Costa et al. 2018): the pronounced increase in air temperature of ~1.26 °C around 1987 triggered a ~40 % increase in sediment concentration in the glacier-fed Rhône river. The intensity of this change is likely to be extremely variable both regionally and seasonally, and can affect water clarity and thus light absorption. The eroded material may also be a source of pollutants that were previously trapped there. Pollutants that had been frozen in the ice would also contribute. However, at a global scale, these pollution sources are significantly smaller than atmospheric deposition and urban sources[3] (N. Chèvre, personal communication).

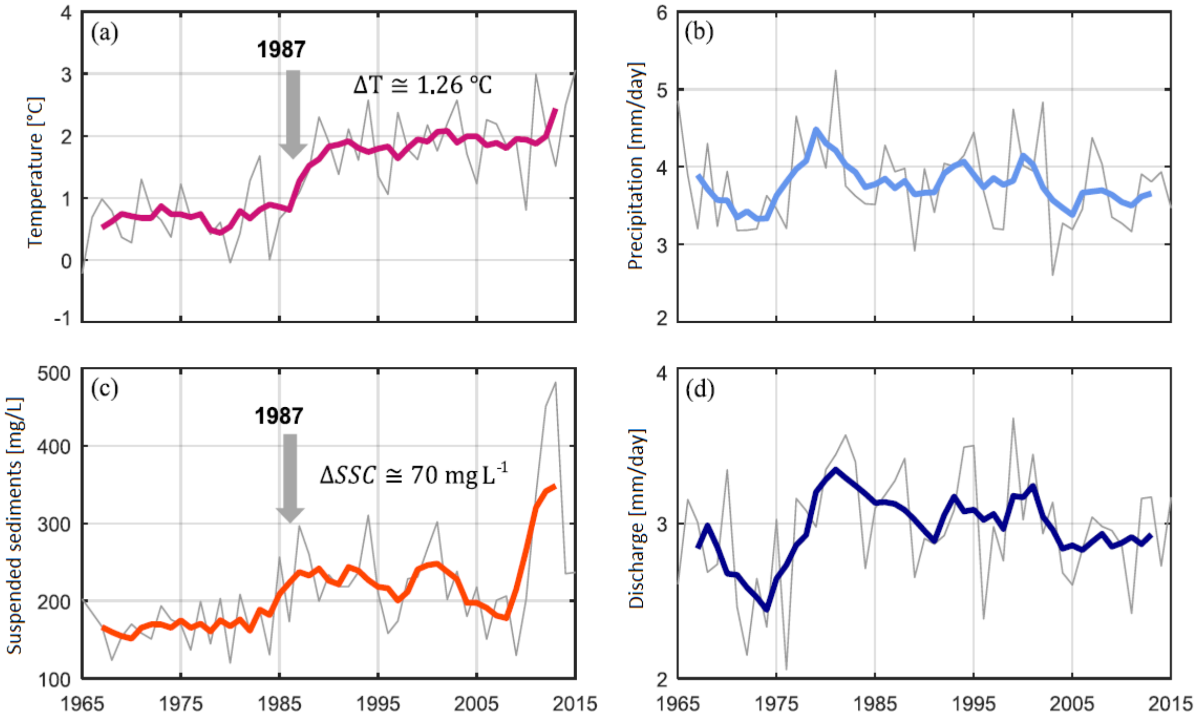


Figure 5.3: Observations for the period 1965–2015 in the Rhône river basin of (a) basin-averaged air temperature, (b) basin-averaged daily precipitation, (c) suspended sediment concentration measured at the outlet of the basin, and (d) daily discharge per unit area measured at the outlet of the basin. Mean annual values are shown in grey and the 5-year moving average is shown with a bold line. The grey arrow indicates the step-like increase in basin-wide mean air temperature. Adapted from Costa et al. (2018).

The extent, depth and seasonality of snow cover determine many dynamics of alpine catchments (Edwards et al. 2007). Snowpack is a significant reserve and source of chemicals and nutrients in such catchments (Hiltbrunner et al. 2005). These chemicals (e.g., nitrate, sulfate, chloride, pollutants) are released preferentially upon melting of the snow, so that early snowmelt is often accompanied by a pulse in meltwater concentrations (Kalbitz et al. 2000). As a consequence, even short snowmelt events can represent a primary source of solutes for an alpine catchment (Sadro et al. 2018) and result in considerable nutrient losses (Edwards et al. 2007). However, the overall effect depends on local hydrology. Intense snowmelt, often observed during the first warm periods in spring or following rain-on-snow events, generates runoff that does not enter the (possibly frozen) soil and can therefore transport large amounts of nutrients directly to surface waters. On the other hand, if snowmelt infiltrates the soil, many nutrients can be taken up by plants (Edwards et al. 2007). In the latter case, a warmer soil can stimulate infiltration and nutrient removal processes (i.e., immobilization by bacteria and plants), which reduced nutrient export during spring in a case study in the United States (Bernal et al. 2012). Due to climate change, snow nutrient exports could (i) increase in winter because of earlier snowmelt and more frequent rain-on-snow events (Morán-Tejeda et al. 2016), but (ii) decrease in the spring because of shorter duration of snow cover and warmer soil.

---

[3] For example for PCBs, an important urban source is leaking from PCB-containing material such as old joints.

However, it is still unclear how the interaction of climate change and snowmelt will affect catchment processes (Musselman et al. 2017). Effects are likely to vary depending on location, altitude, microclimate, and local hydrology.

Warmer temperatures drive the treeline higher up the mountain slopes. A temperature increase of about 4 °C could, in the long term, result in a treeline about 500–700 m higher. This will modify nutrient cycling in the newly colonized soils – an increase (i.e., more nutrient mineralization, uptake and loss) can be expected, unless trees replace previously farmed (and fertilized) surfaces (F. Hagedorn, personal communication). A higher treeline could thereby stimulate nutrient cycling (due to higher inputs) in alpine lakes (Campbell et al. 2000), as well as modulate runoff peaks and reduce erosion of the affected terrain (Lindner et al. 2010). Recent work showed that nutrient availability (and thus leaching potential) increases under newly established trees, and decreases when these nutrients are bound into biomass following further forest expansion (F. Hagedorn, personal communication).

To summarize, alpine catchments will be particularly strongly affected by climate change. The buffering effect of snow cover is likely to diminish in most regions, resulting in drier catchments in summer and autumn. Glacier melting and soil thawing will increase exports of organic and inorganic material, as well as pollutants, to waterbodies. Pollution from alpine catchments will remain very low in comparison to anthropogenically-affected areas (V. Prasuhn and W. Richner, personal communication); however, the changes could be large in relative terms, and thus potentially ecologically relevant.

6 Impacts of climate change on water quality

Box 4: Synthesis – Impacts of climate change on waterbody processes relevant for water quality.

Climate change will affect the physicochemistry of Swiss waterbodies in many complex ways. Higher temperatures will increase oxygen demand in aquatic systems that are not dominated by photosynthesis, and thereby increase the occurrence of anoxia and low oxygen conditions – temporarily in rivers (at night in summer) and river-fed groundwater (during summer droughts), but more and more permanently in the hypolimnion of lowland lakes, especially those where temporary anoxia occurs today.

In rivers, the decrease in dilution capacity during droughts will be a major issue. Below wastewater treatment plants and industrial and agricultural point sources, higher concentrations of contaminants (organic matter, ammonia, phosphate, chlorine, synthetic chemicals, etc.) can occasionally be expected.

In lowland lakes, reduction of deep mixing will be the major driver of declining oxygen concentrations in the hypolimnion. It will also reduce the frequency of winter nutrient replenishment to the epilimnion, which will reduce algal growth and lake productivity throughout the year (if external nutrient inputs are low), but increase nutrient buildup in the hypolimnion (especially phosphorus). Increased temperature will favor cyanobacteria in late summer, but chiefly if nutrient concentrations are sufficient and conditions are rather N-limiting. Summers following a winter with deep mixing, which may result in substantial P upwelling into the epilimnion, will be most critical.

In this section, we review the impacts of climate change on processes occurring within the waterbodies themselves. These impacts are qualitatively summarized in Figure 4.1. Here, we discuss the combined effects of these impacts and of impacts of climate change on catchment processes, which were detailed in the previous section.

Box 5: Trends of organic carbon (OC) in surface waters, and related drivers.

By affecting biochemical processes, climate change may also impact the OC content in waterbodies. OC is a driver of oxygen demand because oxygen is needed to degrade it, and can therefore contribute to the tendency towards oxygen depletion in water. A significant increase in OC concentrations was reported in surface waters in the Northern Hemisphere over the last decades (e.g., Evans et al. 2005, Whitehead et al. 2009), however much of the data comes from humic-rich ecosystems or from headwater rivers (Rodríguez-Murillo et al. 2015). In the main Swiss rivers however, weaker transient trends were observed: an increase in both TOC and DOC from 1974 to 1999, and a more marked decrease from 1999 to 2010 (Rodríguez-Murillo et al. 2015), which contradicts the observations in other systems. The drivers behind these trends are complex and could not be clearly identified; discharge, correlated to rainfall, appeared to be the primary explanatory factor for TOC (Rodríguez-Murillo et al. 2015). In Swiss lakes, Rodríguez-Murillo and Filella (2015) reported a weak increase of OC in large lakes (area > 20 km^2) and a weak decrease in smaller ones. Again, drivers could not be clearly identified; the authors suggested that increased inputs in large lakes, and reduced productivity in small lakes (following re-oligotrophication), could explain the pattern. Air pollution also has an effect. Reductions in atmospheric deposition of nitrate and sulfate, causing catchment deacidification, have been observed to make DOC concentrations rise in mountainous waterbodies (Monteith et al. 2007) and streams in forested catchments (Hruška et al. 2009).

Box 6: Impacts of climate change on hygienic water quality in surface waters.

The hygienic water quality of lakes and rivers is essential for recreational activities such as swimming, and may be impaired by climatic factors. First, high temperatures favor the growth of organisms such as cyanobacteria (see Section 6.2.3), schistosomes, or freshwater jellyfish (Schaffner et al. 2013), in particular in lakes. Such problems will likely occur more often in the future, but their spatial and temporal extent should remain limited – they were rarely observed during the recent hot summer of 2015 (FOEN 2016b). Secondly, strong rain can lead to deterioration of water quality in recipient waterbodies, via inputs of fecal bacteria and many other contaminants, mainly due to combined sewer overflows (see Section 5.3). This problem is well known at specific locations; however, there are no published studies in Switzerland. Still, the combination of more severe droughts (which reduce the dilution capacity of rivers) and stronger rain events (which exceed the capacity of the wastewater treatment plants) will aggravate the hygienic pollution caused by CSOs.

6.1 Rivers

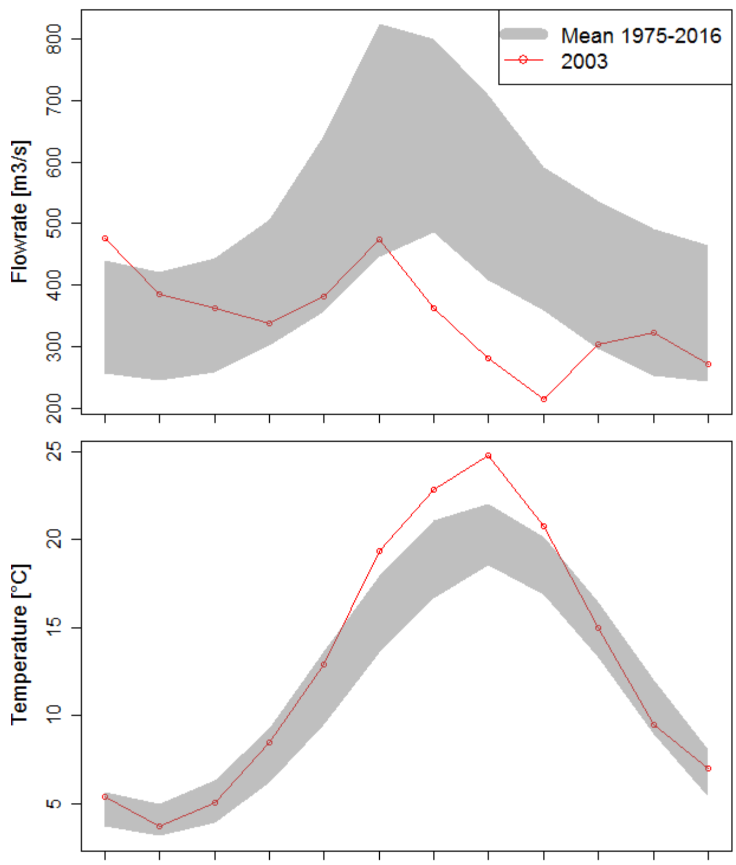
Rivers can be qualitatively split between their upper part (flowing from the source, often through low-impacted areas) and their lower part (flowing through the lowlands with urban and agricultural areas). Except for catchments with a hydropower dam, the dynamics in the upper part depend almost only on natural catchment characteristics, while the lower part can be strongly influenced by human activities. If the lower part is located downstream of a lake, the temporal dynamics (e.g., of discharge and temperature) can be very different, and the river is sensitive to changes in conditions of the lake epilimnion (e.g., trophic status, mineral conditions or oxygen concentrations; Zobrist et al. 2018, Woolway et al. 2018).

6.1.1 Oxygen, redox and pH conditions

Thanks to the high exchange with the atmosphere and to daily photosynthesis, rivers are usually well aerated (oxygen-rich). During summer droughts however, low flowrates and high temperatures interact to decrease the oxygen-carrying capacity of rivers (Ducharne 2008). In eutrophic rivers, this can favor intermittent anoxia: due to increased phytoplankton growth, oxygen demand will be higher during the night respiration phases (Mulholland et al. 1997), and also when the blooms are degraded.

River acidity depends on many factors such as the properties of the catchment soil, groundwater exchange, and atmospheric deposition, so that no general prediction of climate change effects can be made. In the United States, however, it was observed that due to lower flows, the ratio of exported cations to anions increased in the catchment, resulting in stream acidification (Schindler 1997).

Measurement of oxygen and pH in Swiss rivers during the very dry and hot summers of 2003 and 2015 reveal a considerable increase in short-term fluctuations in comparison to other years (Bader et al. 2004, FOEN 2016b). This is due to marked day/night cycles of photosynthesis and respiration. On average, oxygen concentrations were slightly lower in 2003 than during previous years (Figure 6.1), especially in reaches with low turbulence such as smooth, straight channels (Bader et al. 2004). Alkalinity behaved similarly, reaching low levels during the summer 2003 (Figure 6.1), such that the rivers were less buffered against acidification.



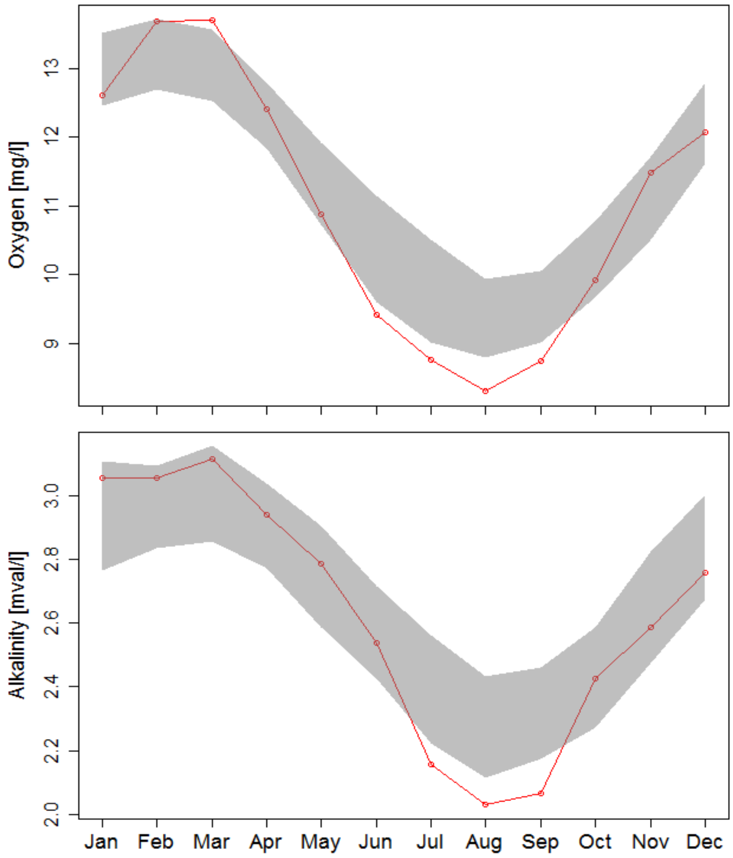


Figure 6.1: Monthly values of (from top to bottom): flowrate, temperature, oxygen, and alkalinity in the Rhine at Rekingen, Switzerland. The grey area shows one standard deviation above and below the monthly means over the whole dataset (1975–2016). The red line shows the monthly means in 2003, a year characterized by summer drought conditions. Data source: NADUF monitoring program (biweekly data).

6.1.2 Carbon, organic matter, nutrients, minerals, xenobiotics

Due to variability of local conditions and complexity of the processes involved, the impacts of climate change on chemical conditions in rivers are difficult to generalize. Most studies have found a positive correlation between flowrate and concentrations of some compounds. For example, increased flowrate has been related to increases in DOC (Evans et al. 2005), DOM (Hejzlar et al. 2003), and, in agricultural catchments, nitrogen (Arheimer et al. 2005). This last study modeled a 7 % DOM increase in central European rivers under climate change scenarios with doubled CO_2 concentrations.

In alpine rivers, reduction of snowmelt inputs may increase spring respiration, thereby reducing OC exports to downstream areas; this was recently observed in an alpine river network in Austria (Ulseth et al. 2018). In permafrost and glaciated watersheds, increased concentrations or DOC, POC, and contaminants can be expected due to increased runoff and erosion of the frozen material (Rouse et al. 1997, Wang et al. 1999, Schindler 2009). Glacier meltwater contains lower concentrations of organic and mineral compounds than lowland rivers, thus increased meltwater inputs in summer would reduce downstream concentrations in rivers fed by large glaciated catchments. For instance, during the hot summer of 2003, the electrical conductivity (representative of dissolved salts) in the Rhône river was about half its normal value (Bader et al. 2004). This, however, only minimally affects water quality.

Warming-induced increase of photosynthesis intensifies uptake of dissolved CO_2, thereby affecting carbonate equilibrium and raising pH (Bader et al. 2004). This results in a higher ratio between toxic ammonia (NH_3) and ammonium (NH_4^+), and potentially in increased nitrification, i.e., transformation of ammonia to nitrate (NO_3^-), with toxic nitrite (NO_2^-) as an intermediate product. These effects are likely to remain limited – they were not an issue during the hot and dry summer 2003 in Switzerland, except downstream of certain wastewater treatment plants (Bader et al. 2004).

In warmer rivers, degradation processes of chemical compounds (such as hydrolysis or biodegradation) accelerates. For instance, faster biodegradation of organic matter (e.g., dead phytoplankton) has been reported (Ducharne 2008). Outside of the growth season, climate change would reduce phytoplankton concentrations, because temperature enhances the loss rate more than the growth rate (Ducharne 2008). In addition to such temperature-controlled biochemical processes, it can be expected that rates of photodegradation of susceptible chemicals such as some pesticides and pharmaceuticals will also increase during summer conditions (warm, sunny, low flow) (Hoffmann et al. 2014).

Intense photosynthesis, which can occur at the beginning of summer if the temperatures suddenly rise under sunny conditions, decreases the concentration of dissolved CO_2. This triggers a reduction in the solubility of calcium, which, in extreme cases, can result in its precipitation. This phenomenon is rare but was reported during the summers 2003 and 2015 (Bader et al. 2004, FOEN 2016b); in the Rhine in early June 2003, calcium precipitation even led to the formation of foam on the water surface.

6.1.3 Processes in the hyporheic zone

The hyporheic zone is the transition zone between surface water and groundwater; permeated with water, it hosts sediment reactions and often controls the exchange between river and groundwater. The hyporheic zone is characterized by high microbial biomass, and hosts many biochemical processes such as reduction/oxidation, dissolution/precipitation, deposition of suspended matter (P, OC, metals), sorption of metals and contaminants, and microbial nutrient uptake. However, the interactions between these important mechanisms and the effects of external drivers are still poorly understood, so that the significance of the hyporheic zone to stream ecology and biogeochemistry is a topic of ongoing research (Boano et al. 2014).

As an example, processes in the hyporheic zone influence river bed clogging (or colmation), which reduces river bed porosity and can block groundwater exchange, thereby altering water quality as well as benthic habitats (Boano et al. 2014). Colmation consists in sedimentation and accumulation of fine particles, followed by chemical precipitation driven by hyporheic exchange (Boano et al. 2014). Colmation is usually reversed by high flows inducing river bed sediment transport (e.g., during floods), and sometimes by bioturbation or by pore water flow (Boano et al. 2014).

As a result of the intense microbial activity, oxygen demand is high in the hyporheic zone. Higher temperatures accelerate oxygen consumption, resulting in lower concentrations in the water returning to the river or flowing into groundwater (Hoffmann et al. 2014), which is ecologically relevant. In the Rhône river in France, a three-day artificial drought rapidly influenced river chemistry as more groundwater flowed in through the hyporheic zone: a decrease of oxygen and nutrient concentrations, pH, and bacterial count was reported (Boissier et al. 1996).

6.1.4 Dilution capacity

In the future, many rivers will experience increasingly severe periods of low flow (above all during summer droughts). As mentioned in Section 3, rivers fed by small non-glaciated catchments are likely to be affected first, while rivers downstream of large (and possibly regulated) lakes will be minimally affected. Lower flow means a lower dilution ratio for any input to the river. Natural sources are often correspondingly low during dry periods; however, anthropogenic sources such as wastewater or industrial discharges may remain constant despite low natural flow. Lower flows therefore lead to higher concentrations of chemicals from such sources in streams (Cruise et al. 1999, Boxall et al. 2009, Hoffmann et al. 2014). As mentioned in Section 5.3, this can have a marked impact in small rivers receiving wastewater inputs of lower quality. At such a location near Basel, high ammonia concentrations up to 0.06 mg NH_3/L were recorded during the very hot and dry summer 2003, while flow rates were extremely low (Bader et al. 2004). In Canton Zürich, dry periods characterized by low flows were also observed to correlate to higher chemical concentrations in small streams – for instance, phosphate concentrations upstream from lakes are significantly higher than under normal conditions (P. Niederhauser, personal communication), hinting towards reduced dilution of wastewater discharges. In the perialpine river Thur at Andelfingen, high chlorine concentrations matched low flowrate during the same summer (Figure 6.2), pointing to reduced dilution of wastewater. In the case of point sources containing nutrients and OC, algal growth and oxygen demand is increased, therefore accentuating the expected direct impacts of climate change (Ducharne 2008). As a counteracting effect, the performance of wastewater treatment plants is usually better during dry periods (Bader et al. 2004), as they receive less water to treat and the risk of overflows (see Section 5.3) is reduced (A. Joss, personal communication).

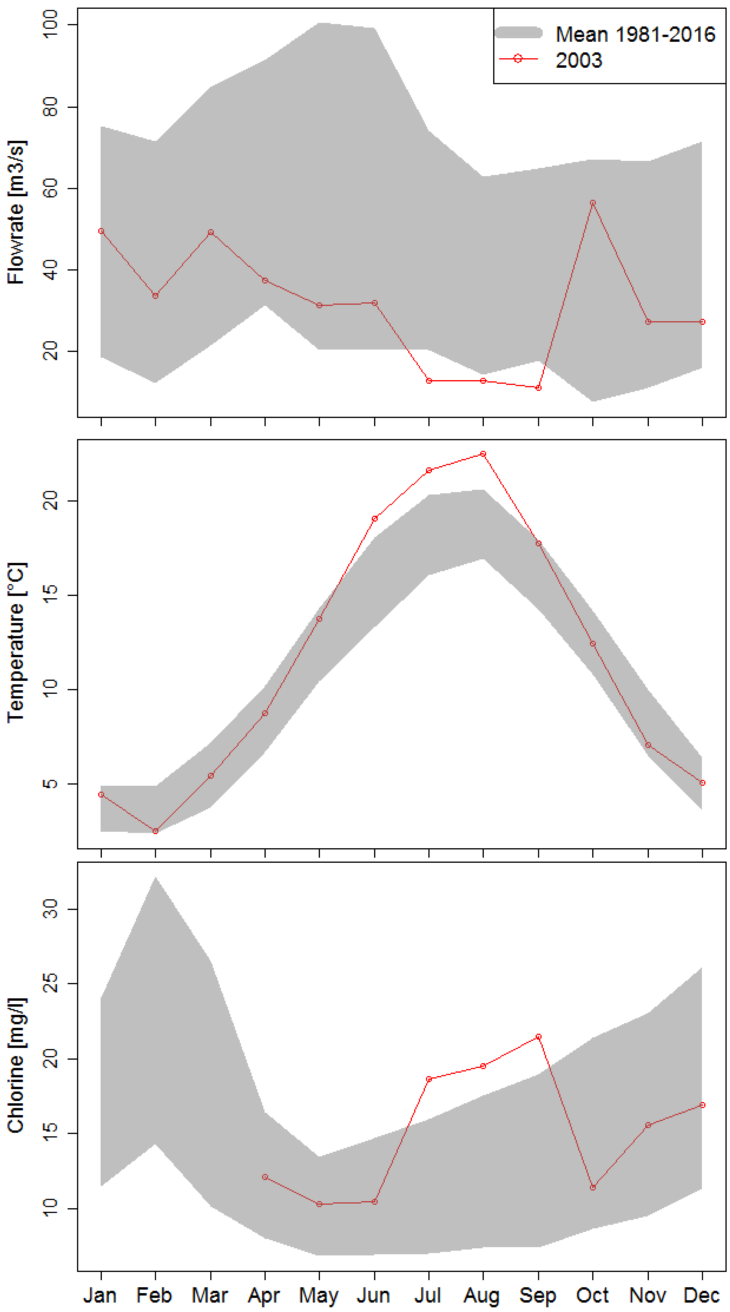


Figure 6.2: Monthly values of flowrate (top), temperature (middle) and chlorine (bottom) in the Thur at Andelfingen, Switzerland. The grey area shows one standard deviation above and below the monthly means over the whole dataset (1981–2016). The red line shows the monthly means in 2003, a year characterized by summer drought conditions. Data source: NADUF monitoring program (biweekly data).

6.2 Lakes

6.2.1 Oxygen, redox and pH conditions

As discussed above (Section 3.2.3), higher air temperature generally increases stratification during the warm season and reduces the frequency and magnitude of deep mixing in temperate lakes. This diminishes oxygen replenishment of the deep water, thus negatively impacting hypolimnetic oxygen concentrations (Hoffmann et al. 2014, Schwefel et al. 2016). This is documented in many large Swiss lakes such as Lake Geneva (Schwefel et al. 2016), Lake Constance (Wahl and Peeters 2014), Lake Zürich (Peeters et al. 2002), Lake Lugano

(CIPAIS 2016a), and Lake Maggiore (CIPAIS 2016b). Anoxia in the hypolimnion modifies the redox conditions, leading to reduction of alternative electron acceptors (e.g., nitrate, sulfate, iron, manganese). Such chemical processes can lead to production of harmful compounds (e.g., nitrite, sulfide), which can considerably deteriorate the water quality of the hypolimnion.

For lakes that inversely stratify in winter, reduced inverse stratification and ice cover would increase vertical mixing during the cold season, favoring oxic conditions in the hypolimnion. The same result is expected from deep inflows which, during winter floods, can bring significant amounts of oxygenated water to the hypolimnion (Råman Vinnå et al. 2018).

In precipitation-dominated lakes, droughts could raise the fraction of groundwater inputs, resulting in higher solute concentrations and possibly acidification (Magnuson et al. 1997). In high alpine lakes, it was reported that increased air temperature raises the water pH, because of alkalinization from faster weathering rates and biological activity (Sommaruga-Wögrath et al. 1997, Koinig et al. 1998, Steingruber and Colombo 2006). A study of 57 high alpine lakes in Austria between 1985 and 1995 revealed (i) a reduction in N concentrations due to increased biological processing, (ii) an increase in sulfate and silica concentrations due to dissolution of minerals (weathering), which is especially intense on newly ice-free surfaces, and (iii) an average pH increase of ~0.25 units (Sommaruga-Wögrath et al. 1997). The effects of these drivers seem to override atmospheric deposition (the main cause of acidification) in these high alpine catchments.

6.2.2 Carbon, organic matter, nutrients, minerals, contaminants

In the future, more temperate lakes may become oligomictic or meromictic (i.e., mixing rarely or never), whereby vertical water exchange is reduced. In such lakes, productivity is usually lower because there is less input of nutrients from the hypolimnion towards the photic layers (Yankova et al. 2017). Figure 6.3 shows the decreasing mixing depth in Lake Zürich over the last 20 years; it appears that epilimnetic P enrichment stopped since the year 2013, a period characterized by warm winters. Figure 6.4 shows the correlation between winter mixing and peaks of P concentrations in the epilimnion in Lake Zug and Lake Zürich. Under conditions of weak mixing, nutrients accumulate in the hypolimnion and, if oxygen becomes depleted, may undergo chemical transformation under anoxic conditions. In some lakes, oxygen depletion leads to increased release of P from the sediments (Jeppesen et al. 2009, Isles et al. 2017). Figure 6.5 gives a conceptual model of the dynamics of the epilimnetic N:P ratio in shallow (< 15 m) and deep (> 15 m) lakes. Climate change is likely to increase N consumption and sediment P release, thereby decreasing this ratio, especially towards the end of summer. Such chemical exchange between water and sediments will also be intensified in warmer hypolimnia (Magnuson et al. 1997). Deep mixing will occur more rarely, but when it occurs, nutrient upwelling is likely to be stronger, thereby triggering worse eutrophication-like responses such as algal blooms in deep lakes (see Section 6.2.3).

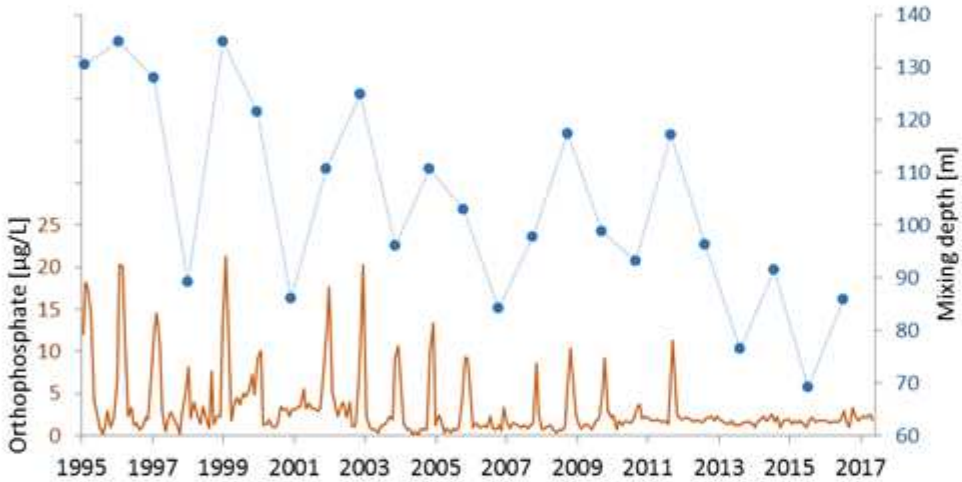


Figure 6.3: Time series of mixing depth and phosphate upwelling in Lake Zürich. Dotted blue line and blue circles: mean vertical mixing depth (right axis). Solid orange line: epilimnetic orthophosphate concentrations (left axis). Adapted from Yankova et al. (2017).

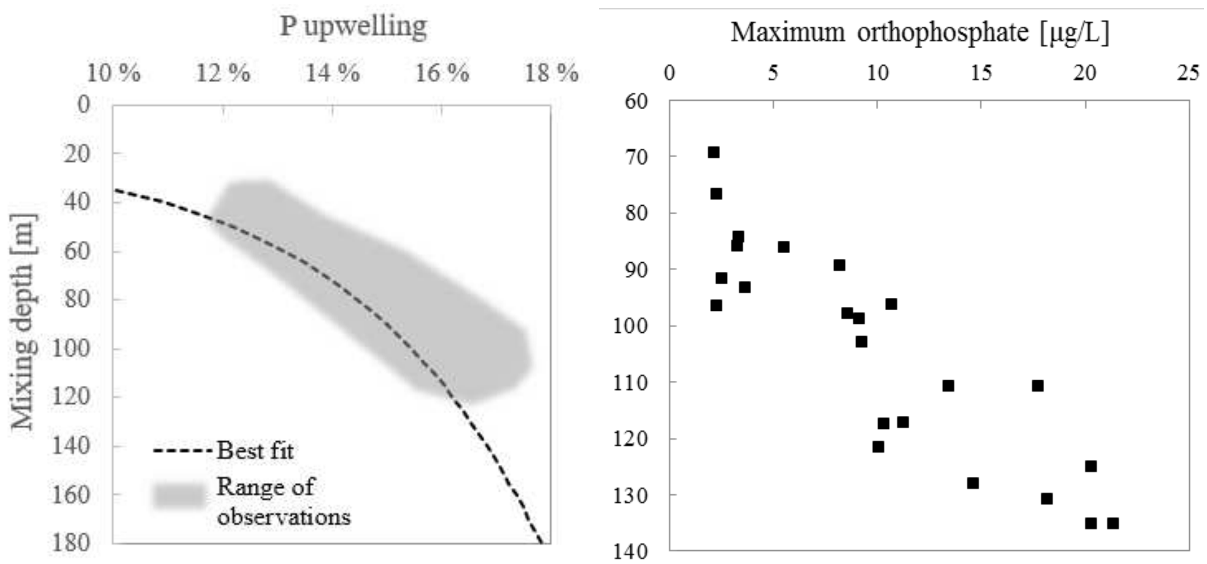


Figure 6.4: Relation between mixing depth and phosphorus upwelling in two Swiss lakes. Left pane: observations and calculated curves for Lake Zug, where each marker represents a year from 1982 to 2015, based on data from Schwefel et al. (submitted). Right pane: observations for Lake Zürich (see Figure 6.3), where each marker represents a year from 1995 to 2017, data source: (Yankova et al. 2017).

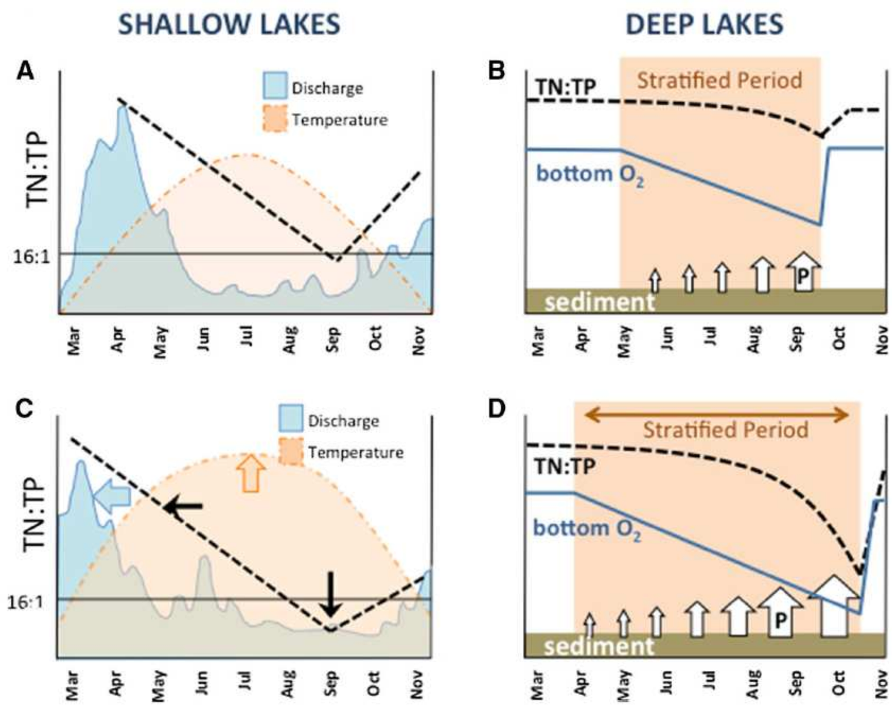


Figure 6.5: Conceptual model of external influences on the ratio of total nitrogen (TN) to total phosphorus (TP) in shallow (< 15 m) and deep (> 15 m) lakes showing intra-annual drivers of nutrient variability (top) and possible responses to changing climate (bottom).
A: Intra-annual patterns of TN:TP in shallow lakes related to tributary inputs and temperature.
B: Seasonal depletion of hypolimnetic oxygen in deep lakes leading to late-season sediment P loading.
C-D: Potential effects of climate change in shallow and deep lakes.
Adapted from Isles et al. (2017).

In lakes, dry periods are characterized by low inflows and high evaporation. During these periods, the water renewal rate is reduced, resulting in accumulation of nutrients and contaminants from point sources (Murdoch et al. 2000, Schindler 2009). Together with temperature increase, this exacerbates eutrophication problems (Arheimer et al. 2005), in particular in lakes under high anthropogenic pressure. Droughts eventually lead to accumulation of persistent compounds and metals but, assuming natural-like inflows (i.e., with a low nutrient load), concentrations of nutrients in the outflow (e.g., N, P, DOC, silica) decrease (Magnuson et al. 1997), because these have had more time to be processed (Schindler 1997, Evans et al. 2005). For instance, higher temperatures were observed to enhance denitrification, an important N sink (Arheimer et al. 2005).

Lake productivity is linked to inputs of dissolved organic matter (DOM) containing nutrients (N, P, etc.). Lakes receiving less DOM are generally less productive, clearer, and with a deeper thermocline (Schindler 2009) – this is the case of most Swiss lakes. DOM inputs generally rise with increasing runoff and presence of wetlands (Schindler 2009). Global warming will increase C cycling and lake productivity in most temperature-limited ecosystems, which are commonly found at higher altitudes or latitudes, or in nutrient-rich areas (Porter et al. 1996, Gudasz et al. 2010). Changes in lake productivity have a feedback effect on light regime (as algae control water clarity), and thus on energy supply, nutrient cycling, and, among others, metal toxicity (Evans et al. 2005). Higher nutrient inputs in winter would cause stronger algal blooms in spring (especially in cold lakes, where nutrients are not used during winter), but should still be less problematic than summer inputs (Arheimer et al. 2005).

In high alpine lakes, winter freezing increases concentrations of chemicals (which do not freeze). In warmer winters, decreased freezing could thus lead to lower chemical concentrations in the lake water (Schindler 1997). In summer, increased drought severity was shown to be a primary driver of high-turbidity inputs to alpine lakes during storms occurring under drought conditions (Perga et al. 2018). These turbid inputs limit light penetration and can alter the physical and biochemical dynamics of the lake for the rest of the summer (Perga et al. 2018). In general, higher sediment loads from alpine catchments have the potential to reduce productivity in recipient lakes.

In addition, a decrease in ice cover (both in thickness and duration) lengthens the growing season and increases epilimnion volume (i.e., deeper thermocline) during the summer. As a result, cycling of chemicals, nutrients and minerals increases (Magnuson et al. 1997, Rouse et al. 1997), which resulted in higher productivity and nutrient concentrations in modeled scenarios (Arheimer et al. 2005). As mentioned above, drier summer conditions would counteract this effect by giving nutrients more time to be processed and eliminated.

Higher temperatures will lead to increased biodegradation of toxins and other contaminants. This would result in lower concentrations of these compounds (Moore et al. 1997), together with increased nutrient uptake and bioaccumulation in sediments (Mulholland et al. 1997).

6.2.3 Phytoplankton growth

Phytoplankton growth in lakes is an important process, as it can strongly affect water quality through changes in clarity, color, taste, smell, chemistry, and toxicity of the water. Phytoplankton growth is dependent on many environmental drivers. Nutrient availability (in particular N and P) is often a primary factor (Isles et al. 2017), and favorable temperature and light conditions promote growth. In summer-stratified lakes, it is typical to observe phytoplankton growth in spring, summer and autumn, sometimes forming blooms if the conditions become non-limiting. Climate warming is expected to globally extend the period of phytoplankton growth and increase its strength in temperature-limited systems such as eutrophic or alpine lakes. For Lake Constance, a 4 °C temperature increase was estimated to trigger the first phytoplankton blooms three weeks earlier, possibly allowing for more blooms throughout the growing season (KLIWA 2015). In addition, hydrological changes such as reduced snowmelt (in alpine catchments) may also favor phytoplankton growth by increasing spring water temperatures and retention time (Sadro et al. 2018).

Cyanobacteria (also referred to as blue-green algae) are generally characterized as well adaptated to higher temperatures (Arheimer et al. 2005), and by the production of toxic substances (cyanotoxins) which can affect water quality (Hoffmann et al. 2014). Cyanotoxin production can be particularly strong and problematic during blooms of cyanobacteria, taking advantage of available nutrients. These blooms are favored by warmer lake surface water (Paerl and Huisman 2008, Wood et al. 2017), both because such temperatures are closer to the organisms' thermal optima and because of the extended growth season. Summer heatwaves were shown to be particularly critical (Jöhnk et al. 2008). Cyanobacteria seem to also benefit from N-limiting conditions (Weyhenmeyer et al. 2007), which can be a result of climate change through (i) faster N consumption during warmer springs, (ii) reduced summer inputs from drier and warmer catchments (Weyhenmeyer et al. 2007), or (iii) P release from the sediments in shallow lakes (Isles et al. 2017). A molar N:P ratio above 15 appears to prevent cyanobacterial blooms (Paerl 2014), although this is not valid for some species such as the non-N_2-fixing *Planktothrix rubescens*, observed in Lake Zürich. Finally, an increase in residence time (which is a result of droughts) allows phytoplankton to remain longer in the water and grow further. Degradation of dying blooms can cause oxygen depletion and consequently have significant ecological consequences (Paerl 2014).

Some effects may counteract the expected increase in algal growth. In temperate lakes, the spring phytoplankton peak may become smaller if fewer nutrients remain available in spring after enhanced winter production (Porter et al. 1996). In deep lakes, increased stratification could reduce vertical mixing and therefore decrease the nutrient supply to the surface layers, thereby limiting phytoplankton growth (Salmaso et al. 2017, Yankova et al. 2017), especially if external nutrient inputs are low. Finally, it was shown that wind is capable of strongly controlling (repressing) cyanobacteria blooms, especially in shallow basins (Isles et al. 2017).

6.3 Groundwater

Groundwater is used as a source of drinking water in many regions of the world, thus its chemical quality is of major interest. In Switzerland, 80 % of the drinking water comes from groundwater (through pumping or natural springs).

6.3.1 Oxygen, organic matter, redox conditions and mineralization

The water quality of river-fed groundwater is partly dependent on the quality and infiltration rate of the corresponding river (Hoffmann et al. 2014). For instance, reduction of oxygen concentrations in groundwater was observed and can be explained by more microbial activity in the river hyporheic zone, which itself resulted from higher temperatures (Hoffmann et al. 2014). Similarly, if there is increased OC in river water, degradation and biological respiration may deplete the oxygen concentration of groundwater (Gurung and Stähli 2014, Hoffmann et al. 2014). Datasets in Switzerland suggest that the governing factor for oxygen depletion in groundwater is temperature (see Figure 6.6) and POM, while DOM plays a minor role (Hoffmann et al. 2014), as its degradation is less influenced by temperature (Gurung and Stähli 2014). Strong flows and high erosion bring more POM and can therefore accelerate oxygen depletion. Low oxygen levels increase the risk of metal dissolution; upon total oxygen depletion, small quantities of toxic nitrite may form due to nitrate reduction. Nitrate acts as a buffering compound for the redox system, i.e., as an oxidizing agent before reduction of metals starts (Hoffmann et al. 2014).

Taking into account the processes described above, oxygen depletion in groundwater can be expected during extreme (hot and dry) summers, as was the case in Germany and Switzerland in summer 2003 (Figura 2013). During high river flows (e.g., floods), colmation is washed away, and water and oxygen levels in groundwater are rapidly replenished (Hoffmann et al. 2014). Therefore, even under conditions of increased temperature and nutrient loading, future situations of constant anoxia are not expected in river-fed groundwater (Figura et al. 2013); however, short periods of oxygen depletion are expected to become more frequent (Figura et al. 2013). More frequent and severe floods would increase the frequency and strength of fast oxygen replenishment (Figura et al. 2013).

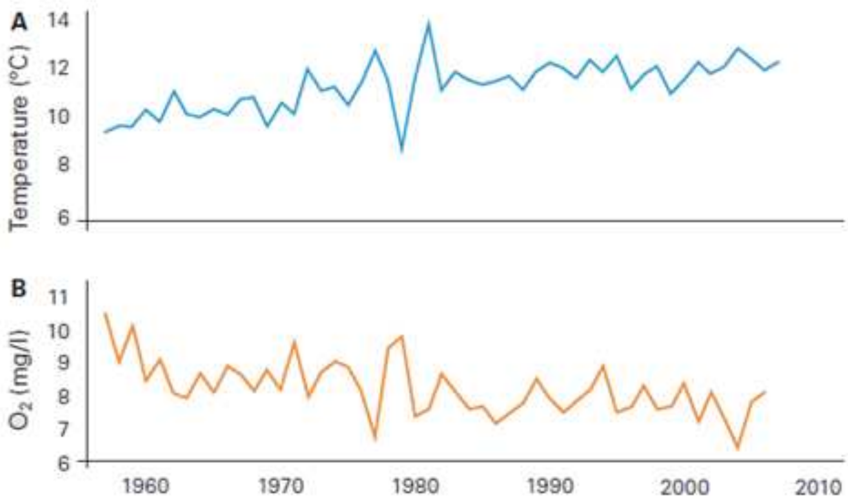


Figure 6.6: Long-term changes in temperature (A) and oxygen concentration (B) in a river-fed aquifer (Seewerben intake, Canton Zürich). Adapted from Kipfer and Livingstone (2008).

The quality of precipitation-fed groundwater depends mostly on the recharge rate, land use, and the properties of the soil through which water flows. Groundwater below polluted soils or nitrate-rich agricultural fields has higher concentrations of these compounds and thus worse water quality than pristine areas. In Switzerland, rain-fed aquifers are not expected to face oxygen depletion problems (U. Von Gunten, personal communication). Nitrate leaching rates into groundwater strongly depend on agricultural practice, in particular the quantity and timing of fertilization and irrigation (Prasuhn and Albisser 2014). In agricultural areas, about 40 % of the NAQUA monitoring sites exceed 25 mg NO_3^-/L, which is the Swiss groundwater quality standard of the water protection ordinance (FOEN 2019).

During dry periods, chemical leaching is usually reduced (Bader et al. 2004) and therefore the input of short-lived contaminants into groundwater is lower (Hoffmann et al. 2014). Due to higher evapotranspiration and reduced nutrient uptake by plants under water stress, substances (e.g., nitrates, DOC) may however accumulate in the soils and can be mobilized and transported into groundwater after abundant rainfall, resulting in a chemical pulse (Evans et al. 2005, Hoffmann et al. 2014). Such pulses can also be caused or enhanced by variations in the groundwater level, which induces washing-off of chemicals from soil that was previously above the water table (Gurung and Stähli 2014). Depending on regional and weather conditions, the effects of such pulses can also last a relatively long time: a strong increase in nitrate concentrations (from 10 to above 40 mg NO_3^-/L) following the drought of 2003 were reported in groundwater near the Thur river for nearly one year (Figure 6.7). According to the results of NAQUA monitoring program, nitrate concentration in Swiss groundwater increased significantly from 2003 to 2004 and from 2005 to 2006 (FOEN 2009). This increase was a result of the very dry year 2003 as well as of the following years with low rainfall. The NAQUA monitoring sites in agricultural areas showed highest increases in nitrate concentration (FOEN 2019). The increase in nitrate concentration in the Thur aquifer was attributed to (i) nitrate enrichment in soil during the dry summer, (ii) fast mineralization of fertilizers during the warm autumn, and (iii) the low water level of the nitrate-poor Thur river, and thus little infiltration of river water into groundwater (Bader et al. 2004).

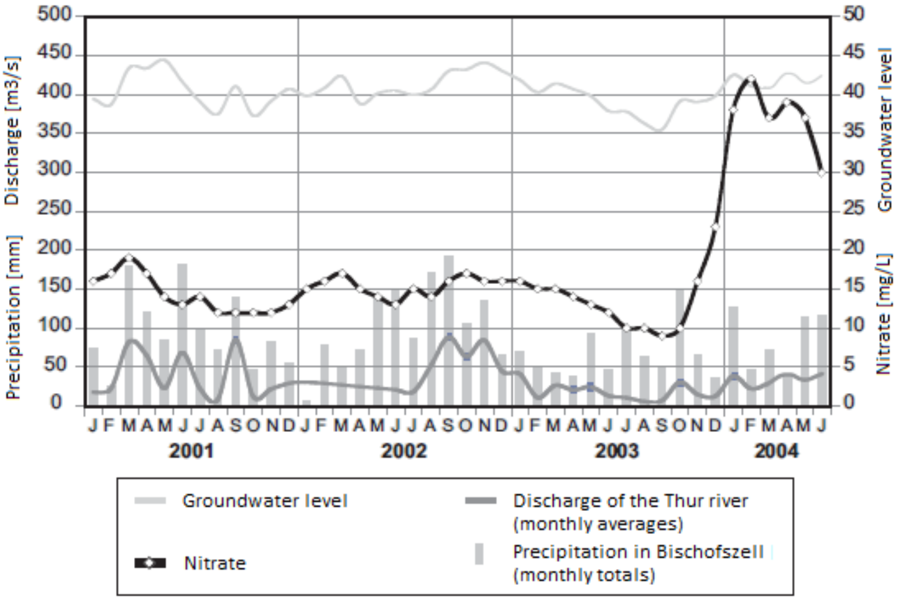


Figure 6.7: Nitrate concentration and groundwater level in a river-fed aquifer (Stocketen bei Niederbüren intake, Canton St. Gallen) from 2001 to 2004, with summer drought conditions in 2003.
Source: Amt für Umweltschutz des Kantons St. Gallen. Adapted from Bader et al. (2004).

An increase in calcium and carbonates (and alkalinity) has been observed in Swiss aquifers over the last four decades, hypothesized result from higher temperatures and increased biological activity in the soil (Zobrist et al. 2018), increasing the partial pressure of CO_2 in the soil (Jeannin et al. 2016). At particular NAQUA monitoring sites, an increase of electrical conductivity was observed during low groundwater levels in summer 2015 (FOEN 2016b). This was attributed to a shift in the composition of the groundwater: there was a higher proportion of older and more mineralized water, due to reduced infiltration of less-mineralized water from recharge by precipitation and river water infiltration. Reductions of chloride and sulfate concentrations were also reported (Hoffmann et al. 2014), although this is likely a result of decreased atmospheric deposition following emission reductions (Monteith et al. 2007). These changes will likely remain unimportant concerning groundwater quality.

6.3.2 Input and degradation of xenobiotics

The soil acts as a buffer zone for compounds flowing into groundwater, although transport can be rapid through preferential flowpaths (e.g., deep cracks, wormholes, root holes, stony and sandy soil). As a consequence, primarily persistent and mobile (weakly-sorptive) substances make it into groundwater (Walter and Hänni 2018). It can be expected that inputs will increase in wetter winters, and decrease during drier summers. Degradation of persistent chemicals will be favored by climate warming.

The results of xenobiotics measurements in Swiss aquifers (NAQUA) for the period 2004-2006 were compiled by FOEN (2009), considering many classes of compounds.

7 Effects on aquatic habitats and aquatic ecosystems

Box 7: Overarching conclusions – Impacts of climate change on freshwater ecosystems.

Climate change is already impacting the ecological status of aquatic ecosystems and aquatic habitats in Switzerland. Climate-associated changes in processes and states in freshwater ecosystems have already been observed for almost every organismal group we investigated in the report. The causes of the changes are diverse and interactive, but in some cases one single parameter is a dominant driver (e.g., temperature change, lake stratification, emergence of diseases).

Climate change is expected to further change many aspects of aquatic ecosystems, however due to complex interactions between environmental factors and uncertainties in their prediction, these changes are difficult to predict.

The most predominant change in the state of freshwater ecosystems is the increasing number of species, especially at higher elevations. However, this increase often go hand in hand with a strong homogenization of the aquatic community (loss of specialist species to generalist ones), disappearance of cold-adapted species, and a loss of genetic diversity. Warm-adapted species, likely including many non-native species, will profit and substantially change the structure of aquatic communities. Furthermore, there will be a shift towards disturbance- and drought-adapted species. Diseases of economically important organisms, such as fish, will likely increase due to warmer temperatures and the prolonged summer season. In parallel, increases in temperature and changes in lake stratification will promote the occurrence of unwanted cyanobacterial growth and blooms. Climate change will also affect phenology, resulting generally in earlier biological cycles and longer growing seasons. How such biological changes interact with altered seasonal flow regimes to cause further ecological change remains unclear.

Overall, aquatic ecosystems will become more similar to each other, lose their local community identity (in particular the specialist cold-adapted and higher elevation species), and become more prone to dominance dynamics by unwanted species (invasive species, diseases, cyanobacterial blooms). Extreme heat events will lead to peaks in water temperature and increase stream intermittency, both of which will cause increased mass mortality events of aquatic organisms. Increases in flood severity will have similar consequences, and will influence the sediment regime in alpine catchments and therefore water turbidity, which can radically affect ecosystem functioning. Generally speaking, most of the expected climate-driven changes can also be reinforced by habitat changes and pollution.

In Sections 5 and 6, we characterized expected changes in aquatic systems from an abiotic perspective, focusing on physicochemical changes as a direct result of climate change, such as changes in temperature, runoff, intermittency, and homogenization of aquatic systems. We will now focus on biotic changes in aquatic systems, including effects on habitats (this section), communities, and organisms (subsequent sections). Aquatic systems are the habitat for many biota (Rolls et al. 2017), and effects of climate change on aquatic systems will have large effects on organisms living there. Thus, climate change will affect aquatic ecosystems, habitats, and the organisms themselves. Here, we will focus on surface waters, for which we have most knowledge. Much less is known on groundwater ecosystems: the first Swiss overviews of microbial groundwater communities were only established by Hunkeler et al. (2006) and Kötzsch and Sinreich (2014), and in both these studies, the state and predicted changes are less clear than for freshwater ecosystems.

Climate change has two major effects on habitats and ecosystems. First, it will directly affect local physicochemical and general abiotic conditions (see previous sections). It will thus affect the organisms' ability to persist locally, such that their "ecological niche" is no longer matched by the local conditions. Secondly, climate change will alter the connectivity and topology of freshwater ecosystems, especially in river networks. For example, individual stretches of rivers will dry up (so-called intermittent rivers), and previously connected river sections will become fragmented. This can happen in both small and large rivers, as for example commonly observed in the Töss river. Desiccation of individual river reaches results, at least temporarily, in increased fragmentation.

The connectivity and topology of freshwater systems are key attributes for understanding diversity patterns (Altermatt et al. 2013), as they define how species can migrate between sites and habitats (Pandit et al. 2017). Considering these properties is therefore essential to better predict the impact of climate change: connectivity will define how species are able to (spatially) track their local environmental optima as climate change is shifting habitats. Indeed, to overcome the temperature increase resulting from climate change, species could shift their altitude range to reach a habitat corresponding to their thermal preferences (Isaak and Rieman 2013). However, this is only possible when the other ecological requirements of the species are fulfilled; for example, a shift in the stream reach may allow species to track a thermal niche, but if other elements of the habitat, such as river bed structure or velocity, do not match the species' ecological requirements, they may not be able to use this new habitat.

Due to its specific structure, the freshwater network is often isolated and/or fragmented within the terrestrial landscape (Woodward et al. 2010a). Several barriers further affect the connectivity of the freshwater ecosystem (e.g., land/water interface, watershed divides, in-stream discontinuities, dams, or warmer temperature), thus reducing organisms' ability to colonize new habitats (Isaak and Rieman 2013). Moreover, streams can be linearly distributed across an altitudinal gradient, thereby reducing movement towards higher elevation (Cianfrani et al. 2015). Even though connectivity allows species to migrate, it does not necessarily imply that the species will reach a suitable lower-temperature area. For example, the distance to cover might exceed species' dispersal capacity (Angermeier 1995).

Another limitation for the potential to track habitat conditions under climate change by shifting ranges is that altitude and habitats are not independent. A shift to a higher-elevation habitat generally translates into other changes in habitat properties (e.g., slope of the stream, gravel size spectrum, streamflow, disturbance regimes) that will not be similar, even though the water temperature may be (J. Brodersen, personal communication).

Thus, not all species and habitats can (and will) move to higher elevations and track their climate niche under climate change, and some habitats (e.g., slow flowing, cold streams) simply do not exist at higher elevations. Studies have already shown that some species cannot follow the temperature change related with climate change and, consequently, experience a range contraction and net loss of habitat (Figure 7.1; Comte and Grenouillet 2013, Eby et al. 2014). While this has hitherto been shown for a few case studies, it may be a generality for most of the (less mobile) native aquatic species in Switzerland.

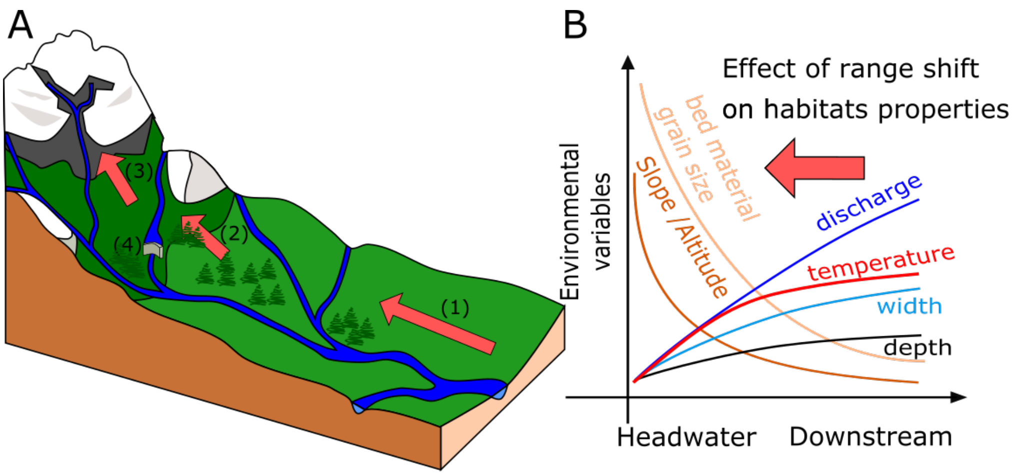


Figure 7.1: A) Conceptual diagram representing the consequences of range shifts for species, B) Changes of habitat properties according to the position in the catchment.

(1) At lower elevation, the distance to cover in order to access elevational refugia can exceed the dispersal abilities of the organisms. (2) Mountains can be elevational refugia. (3) Shifting to higher elevation is not always possible as they may lack refugia or other habitat characteristics change with elevation. (4) Barriers can prevent organisms to reach the elevational refugia.

8 Effects on specific species/species groups/biodiversity

Some of the most relevant possible impacts of climate change on the ecological status of freshwater ecosystems are effects on specific species, species groups, and biodiversity as a whole. The reason for this is twofold.

First, both the individual species as well as their sum (i.e., communities and biodiversity of ecological systems) are the backbone of the stability, resilience[4] and functionality of ecological systems (Cardinale et al. 2012, Loreau and de Mazancourt 2013, Urban et al. 2016). Any climate change related change/shift of species' occurrence and abundance will thus directly affect the stability, resilience and functionality of ecological systems. As all species have generally well-defined ecological niches, they may be directly affected by climate-induced changes in abiotic conditions (see Sections 5 and 6). Specifically, expected changes in water temperature, desiccation and runoff regimes, and changes in the chemical composition of the water due to land use changes associated with climate change, are often directly associated with key elements of species' ecological niche, and thus expected to have direct and immediate effects on species occurrence. The effects will be especially noted for species/species groups that are highly visible or of direct relevance for humans, such as fish (recreation and fisheries) or cyanobacteria (toxic algal blooms in lakes), and many of the expected climate-induced consequences on aquatic ecosystems are best communicated with individual species as examples: while a change in the mean annual temperature or the stratification of a lake is often abstract, the local extinction of fish (such as brown trout) in streams due to heat waves or the occurrence of algal bloom due to warming of lakes are very direct and obvious changes, which can be used as proxies to illustrate climate change effects on aquatic ecosystems.

Second, the occurrence of individual species and biodiversity as a whole is one of the most important variables used to indicate the ecological status of ecosystems, and changes in these

[4] Resilience is the capacity of an ecosystem to return to its former state after a perturbation.

structures may affect interpretation of the monitoring data. For example, in Switzerland, biodiversity and the occurrence of specific species are used in the monitoring of freshwater ecosystems at communal, cantonal and national levels. Most important are probably the nationwide monitoring systems “NAWA” and the Swiss biodiversity monitoring “BDM” (FOEN 2013, BDM Coordination Office 2014, Kunz et al. 2016). These monitoring programs are designed to track changes in biodiversity in order to have an early warning tool for ecological problems, but also a policymaking tool with respect to biodiversity protection. However, they are also used to derive indicators describing ecological status, for example the water quality status or the assessment of river dynamics and structure (FOEN 2013, BDM Coordination Office 2014). Thus, changes in species’ occurrence and biodiversity due to climate change can, on the one hand, be directly tracked by such monitoring. On the other hand, however, changes due to further factors (land-use changes etc.) may mask climate change driven dynamics, or could even affect the suitability and usability of the indicators and indices developed. For example, the assessment of macrozoobenthos in the Modul-Stufen-Konzept (MSK; Stucki 2010) is based on the occurrence of a list of indicator taxa. If the occurrence of these taxa changes, it is then interpreted as changes in the ecological dynamics and structure, such as the river bed structure, even though the change could be due to climate warming, and would then lead to potentially incorrect conclusions. Also, from a methodological perspective, this monitoring has been developed based on current climatic and phenological settings (Stucki 2010), but may need to be adjusted due to climate warming: for example, the appropriate time window for the sampling, or the elevational stratification, may no longer be valid under future climatic conditions. In order to guarantee long-term comparability of bio-indices (e.g., IBCH) based on biodiversity monitoring data, such methodological adjustments need to be considered (see also Urban et al. 2016; Monchamp et al. 2017); however, such an evaluation is beyond the scope of this report.

Changes in species occurrence and biodiversity (mostly declines) are two of the most obvious changes in ecosystems, and are especially pronounced in aquatic ecosystems. Biodiversity loss and the change in species composition due to, for example, non-native species, is stronger in freshwater than in other ecosystems (Dudgeon et al. 2006, Leuven et al. 2009, Vörösmarty et al. 2010). Most of these changes are due to a combination of factors, including habitat change and pollution, but also climate change. Disentangling the individual contributions of these factors is often very difficult. Thus any change in species occurrence and biodiversity must be considered in the context of these multiple drivers, making better forecasts of biodiversity changes under climate change very complex (Urban et al. 2016).

Importantly, a large proportion of past studies on changes in ecological status and diversity of freshwater ecosystems has focused on local environmental or anthropogenic drivers. Diversity patterns in aquatic ecosystems, however, are also strongly driven by the network, with higher diversity levels found in the downstream area and lower diversity in headwaters (Figure 8.1; Altermatt et al. 2013, Alther and Altermatt 2018). This pattern has strong implications in the context of extinctions driven by climate change, for example when the system’s network structure changes due to physical changes (damming, desiccation of river sections), or when climate-induced warming makes the system more permeable for invasive species. For any evaluation of effects of climate change on species richness and biodiversity, an integrated approach is thus needed, considering both local effects and spatial dynamics.

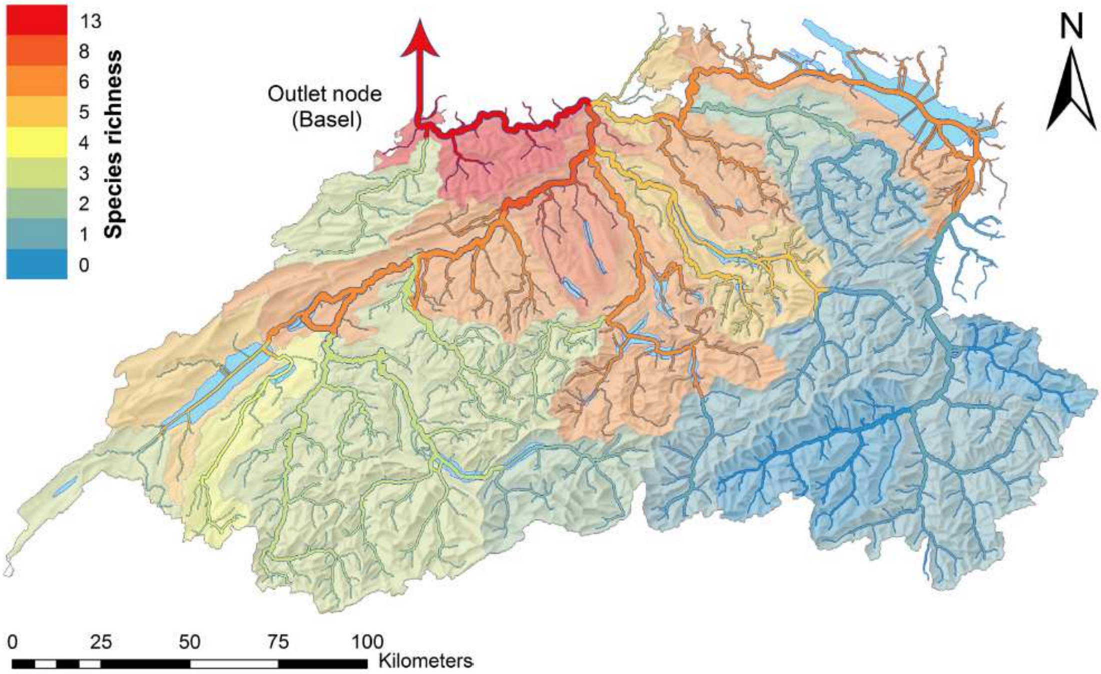


Figure 8.1: Illustration of the diversity patterns in dendritic networks. Amphipod species richness in the river Rhine drainage basin of Switzerland. Local species richness along the fluvial network is depicted as a heat map, with streams and underlying catchments colorized with respect to the observed catchment-level species richness. From Alther and Altermatt (2018).

8.1 Effects on richness and composition

Climate change will affect general diversity patterns by homogenizing the composition of communities (Urban et al. 2016). It can both increase but also decrease richness, depending on the local context (e.g., lake vs. stream ecosystems) and the organismal group considered, and will likely have different effects for low versus high elevation freshwater ecosystems. We first discuss overall effects, and then in subsequent subsections discuss effects on specific taxonomic groups that are relevant either from an economic and policymaking perspective (e.g., fish) or are used in monitoring programs, such as NAWA (national monitoring of the surface water) or BDM (Biodiversity monitoring).

At high elevations, we expect warmer temperatures to decrease the contribution of snow and glacial meltwater to the runoff. As a consequence, the conditions become less extreme, and α-diversity (i.e., local richness) should increase as more generalist species can colonize new habitats (Brown et al. 2007). This increase in richness is expected to be slow, and richness may take decades to adjust to the new environmental conditions resulting from climate change (Menéndez et al. 2006). However, in some alpine streams, a rapid shift in richness and composition of aquatic invertebrates has already been found, with both the number of species increasing and the communities becoming overall more similar due to the loss of specialist species (Robinson et al. 2014). This pattern has been found both in streams created after recession of glaciers, as well as due to changes in runoff regimes associated with hydropower. Thus, at high elevations, richness can rapidly increase and composition can change with similar speed, generally through loss of specialist species and homogenization.

In contrast, the disappearance of extreme habitats is predicted to reduce habitat heterogeneity and thereby decrease β-diversity (i.e., among-community differentiation) (Brown et al. 2007, Jacobsen et al. 2012). Rolls et al. (2017) reviewed the effect of hydrological changes (such as those expected with climate change) on biodiversity at the regional scale (γ-diversity), local scale (α-diversity) and between-site scale (β-diversity). Runoff from glacier melting will result in higher connectivity, for example in flood plains, and is expected to increase general taxon richness. However, this pattern will subsequently reverse as glaciers shrink, and α-diversity will decrease. For functional richness, patterns are unclear as highest values are observed for intermediate values of connectivity. β-diversity, however, increases with fragmentation so that we expect the opposite pattern to the one for α-diversity: a decrease followed by an increase of β-diversity (Figure 8.2). Effects of climate change on γ-diversity (that is, richness at the level of complete catchments/Switzerland) are hard to predict, as these will depend on whether and how long local specialist species will be able to persist, and also how quickly and how extensively novel, hitherto non-native, species arrive. Changes in γ-diversity are expected to have relatively long time-lags with respect to the environmental change driving them.

At low elevations, we expect a different effect of climate change on biodiversity. For small- to medium-sized rivers, we expect higher intermittency (i.e., desiccation) in summer due to decreasing precipitation and less meltwater from the glaciers. Such changes and/or reductions of the hydroperiod (i.e., the time period during which the river contains water) generally reduce local species richness (Soria et al. 2017), thus intermittent streams/rivers have on average lower α-diversity than their perennial counterparts, and communities are generally also more similar. For larger streams and lakes, the change in diversity is unclear. For example, in perialpine, mostly low-land lakes in Switzerland and Northern Italy (Monchamp et al. 2017), an increase of diversity in phytoplankton has been observed, resulting from a combination of warming and environmental change (re-oligotrophication).

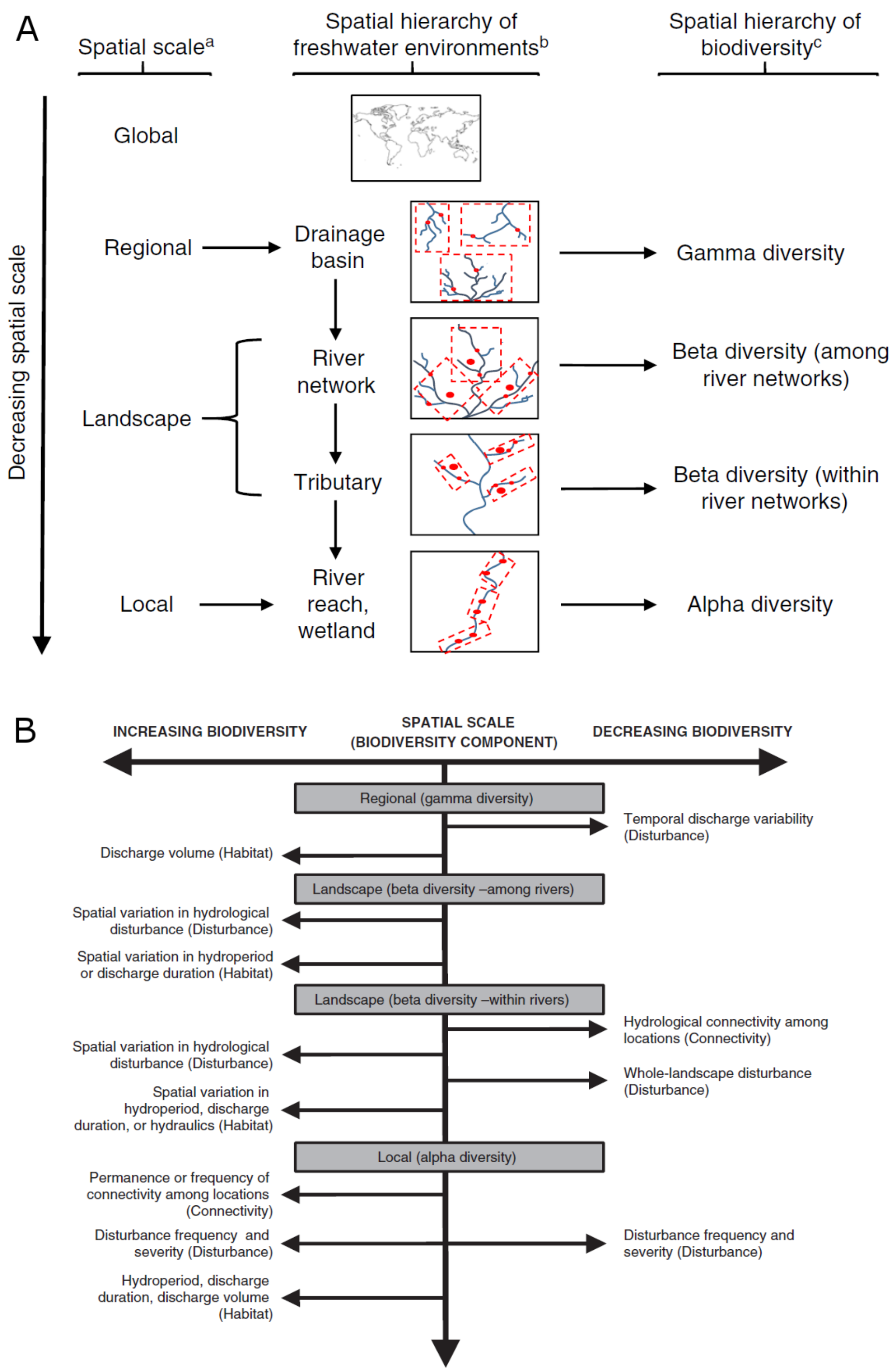


Figure 8.2: Effect of changes in hydrological regime at different spatial scales. A) Schematic diagram representing how riverine environments (including floodplains and wetlands) are arranged as a nested hierarchy of spatial scales and integrated into analyzing and understanding freshwater biodiversity across spatial scales. This diagram integrates common terminology of spatial scales adopted in the ecological literature ([a](Whittaker et al. 2001)), conceptualizations of the hierarchical nature of river-floodplain systems ([b](Frissell et al. 1986)) and applicable measures of biodiversity relevant to each level within river-floodplain systems ([c](Pavoine et al. 2016)). B) Conceptual model summarizing patterns of freshwater biodiversity across multiple spatial scales along distinct ecohydrological gradients in surface water and floodplain environments. The arrow direction along the gradient of increasing-decreasing biodiversity indicates the effect of each ecohydrological mechanism, for each spatial scale of biodiversity. For example, increasing temporal discharge variability leads to decreasing regional biodiversity. Other variables unrelated (or only indirectly related) to hydrology (e.g., latitude, geomorphology, catchment characteristics) are not included but recognized as relevant to structuring biodiversity across spatial scales. The positive and negative effects of hydrological disturbance regime and local-scale biodiversity reflect contradictory and hump-shaped responses reported in the literature. Adapted from Rolls et al. (2017).

8.1.1 Fish

The structure and diversity of fish communities in Swiss lakes and (to a lesser degree) streams are relatively well studied (Vonlanthen et al. 2012, FOEN 2016c), for example through projects such as “Projet Lac”. Besides climate change, important anthropogenic drivers of fish diversity are habitat modification, damming and hydropower use, eutrophication, and non-native species, while less is known about the impacts and relevance of climate change.

The brown trout (*Salmo trutta*) is the most studied species among freshwater fish in Switzerland, and this is also true regarding the effects of climate change (Hari et al. 2006, Cianfrani et al. 2015, Junker et al. 2015, Carraro et al. 2017). It can be used as a surrogate species, since possible effects of climate change on brown trout are likely to be equally relevant for other fish inhabiting cold, oxygen-rich streams and rivers. Brown trout is an important wild fish and is also farmed for hobby anglers (Hari et al. 2006), but its population has experienced a strong decline (Borsuk et al. 2006). Overall, warmer temperatures are detrimental for the species’ survival and reproduction. Temperature influences the onset of spawning (Wedekind and Küng 2010), sex determination (Craig et al. 1996, Baroiller et al. 2009; but see Pompini et al. 2013 for a different finding), fry development, prey availability (Crozier et al. 2008), and the dynamics of diseases such as proliferative kidney disease (PKD) (Carraro et al. 2017). Increasing water temperatures associated with climate change will generally be negative for all of these aspects. Temperature changes are likely to have a stronger impact on populations in the Swiss Plateau, where brown trout lives closer to the upper limit of their temperature tolerance range compared to populations at higher elevations. Warming is likely to also favor competing trout species, such as graylings *Thymallus thymallus* (Hari et al. 2006). However, brown trout and graylings do not naturally compete strongly with each other, because they typically live in different zones. Only because of its relevance for fisheries has brown trout been widely introduced to regions where grayling is naturally the dominant species, and where such competition may then occur.

The optimum temperature range for brown trout’s complete life cycle is between 8 and 19 °C (Burkhardt-Holm et al. 2002), depending on population and life stage. Climate change also increases the risk that water temperature reaches the lethal temperature for brown trout (25 °C) (Hari et al. 2006), which could therefore necessitate a range shift to higher altitudes in the upper parts of rivers. However, the large number of barriers along rivers are likely to prevent the fish from reaching potential refugia. Consequently, the upward shift of habitats would lead to a range reduction and further population decline (Hari et al. 2006).

More severe and frequent floods in winter will make the conditions less favorable for the brown trout’s incubation (Junker et al. 2015). This effect should be less pronounced in natural streams where flood power is buffered and the habitat diversity is maintained. In contrast, in embanked or artificially channelized watercourses, flow power is increased and frequently leads to a predominance of high-energy habitats where the eggs can be scoured and juveniles displaced (Goode et al. 2013, Hauer et al. 2013). Small tributaries may become more important for brown trout due to reduced discharge and lower flood power compared to main river channels (Junker et al. 2015). Considering this possible change, longitudinal and lateral connectivity is of particular importance, and removing dams to grant access to these streams may mitigate the effect of climate change (Junker et al. 2015). While dam removal has barely happened in Switzerland, it has become a relatively common management action in North America in recent years, even for large dams (e.g., Bellmore et al. 2017).

Climate change also affects aquatic populations of organisms through changes in host-pathogen interactions (Engering et al. 2013). One of the main causes of the decline of brown trout is PKD (Borsuk et al. 2006). The pathogen responsible for this disease is the myxozoan endoparasite *Tetracapsuloides bryosalmonae*. The primary hosts of the pathogen are freshwater bryozoans (Hedrick et al. 1993, Canning et al. 2000). Parasite spores are released into the water during an overt virulent phase and infect salmonids through contact with the skin or the gills (Hartikainen and Okamura 2015). Bryozoans are infected by spores released through the urine of infected fishes (Hedrick et al. 2004). In farmed-fish population, mortality rates range from 20 to 95 %. Mortality in wild populations, however, is poorly documented. This disease is very sensitive to temperature, as symptoms only appear when the temperature is above 15 °C and mortality greatly increases with rising temperatures. Carraro et al. (2017) proposed a method to investigate these complex interactions between host and pathogens in the context of climate change to better understand this phenomenon.

Compared to these complex and intertwined effects of climate change on brown trout as a stream-inhabiting fish, not much is known on effects of lake-dwelling fish. We speculate that they will be both directly affected by warming of the water, and indirectly affected by impacts of climate change on plankton dynamics (e.g., Monchamp et al. 2017). Clearly, further research is needed for better understanding the effects of climate change on lake fish.

A further, more commonly observed effect of climate change on fish is the more severe and frequent droughts during summer when streams become desiccated (e.g., very pronounced in 2018). Such droughts and stream intermittency are relatively common in smaller streams and some larger rivers due to their specific hydrology (e.g., Töss), and have been especially pronounced in years with prolonged dry periods, heat waves, and low precipitation (e.g., 2003 and 2018).

8.1.2 Macrozoobenthos

The diversity of stream macroinvertebrates in Switzerland is highly linked to the country's topology (altitude, slope and hydrological network topology) (Altermatt et al. 2013, Wigger et al. 2015), water temperature (Milner et al. 2001, Malard et al. 2003), human land use and chemical pollution (Kaelin and Altermatt 2016), and discharge (Fuente et al. 2017). Alpine floodplain rivers are heterogeneous in terms of physicochemical composition and these differences constitute important drivers of macroinvertebrate diversity (Figure 8.3; Burgherr and Ward 2001, Brown et al. 2003, Brown et al. 2015). Climate change is expected to drive the environmental conditions towards more homogeneous habitats, and would therefore modify these patterns of macroinvertebrate diversity to a more uniform distribution of species, also favoring warm-tolerant species. Macroinvertebrates respond quickly to changes in climate; inter-annual variations in temperature already have a strong impact on macroinvertebrate communities (Rüegg and Robinson 2004). Both temperature and runoff will be affected by climate change, so we expect relatively strong changes in the invertebrate communities, both with respect to diversity as well as composition (Jacobsen et al. 2014). Since Switzerland has a large mountainous area and many alpine species, the effect of climate change on high-elevation species is among the most relevant impacts, also when considering the country's responsibility for managing these species at an international level. In the following paragraphs, we thus focus specifically on climate warming effects on subalpine and alpine aquatic systems.

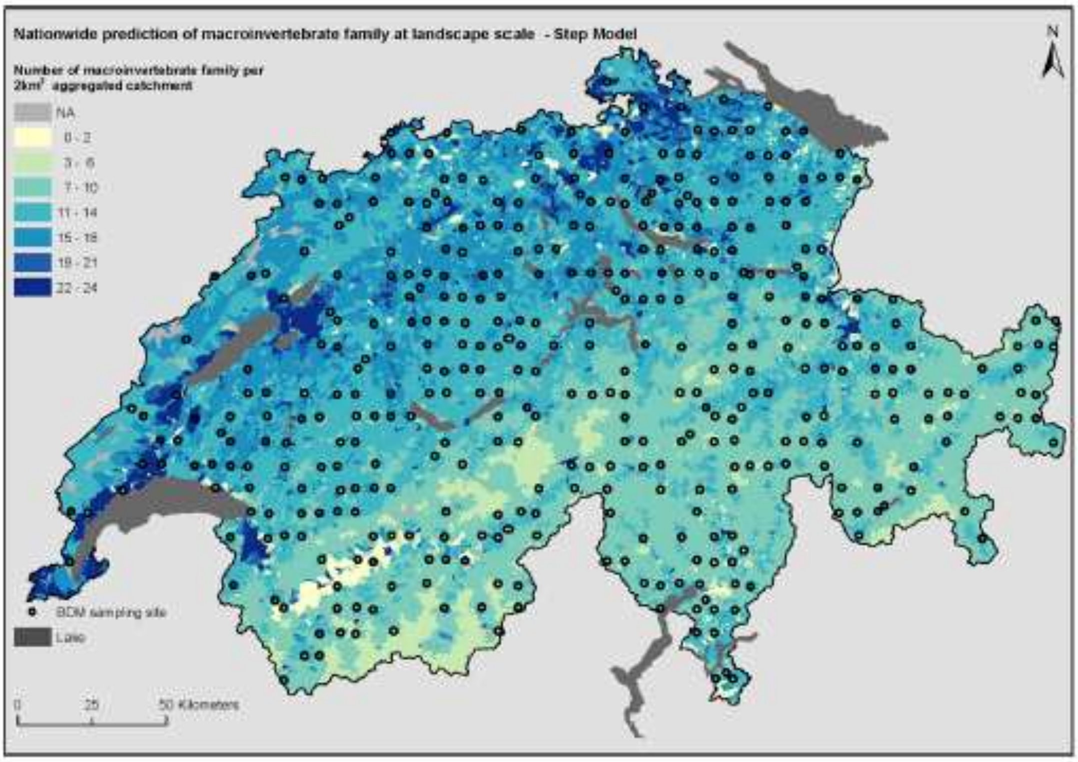


Figure 8.3: Predicted macroinvertebrate family richness across the whole of Switzerland for aggregated catchments at the 2 km² level. No predictions were made for lakes (dark gray areas). EZGG-CH (FOEN), Vector 25 (swisstopo), BDM Sites (Koordinationsstelle BDM). From Kaelin and Altermatt (2016).

Rosset and Oertli (2011) described the 'winners' and 'losers' of warming in lentic freshwater systems in Switzerland. According to their study, 11 % of Coleoptera (beetles, 13 species) and 33 % of Odonata species (dragonflies, 18 species) face an increased extinction risk based on their thermal preferences. 63 % of Odonata species (33 species) are potential winners.

The species identified as winners occur mainly in the lowlands and would shift their range to colonize new habitats (Rosset and Oertli 2011). Eight of the 10 least resilient Odonata were identified as at risk due to their low resilience to temperature perturbation (Rosset and Oertli 2011). In a recent study, Paillex et al. (2017) investigated the influence of a thermal gradient and connectivity in a large floodplain on macroinvertebrates. Connectivity had a greater effect on macroinvertebrate richness than the temperature gradient. However, temperature had a significant effect on the community composition, and the highest densities of non-native species were found at the warmer sites (which, however, are generally also the most connected sites, such that effects of climate and connectivity on invasive species are often hard to disentangle).

Warmer future water temperatures may increase pressure on native macroinvertebrates. Invasive macroinvertebrates could strengthen their impact on native species through competition for resources and habitats, and alter the food web structure (Gallardo et al. 2016, Paillex et al. 2017). Sertić Perić et al. (2015) studied streams where glacial influence has decreased, and predicted an increase in the density of more generalist taxa. According to Ilg and Castella (2006) and Robinson et al. (2014), density and diversity are lowest near the glacier and can be up to 10 times higher 1 km downstream. In glaciated areas, community composition has a strongly seasonal component; winter runoff regimes are more uniform than summer, when there is higher variability due to the variability of the floodplains and to the higher proportion of groundwater that composes the runoff (Brown et al. 2015).

Changes in macroinvertebrate communities due to the lengthening of streams in areas where glaciers are shrinking (due to climate change) have already been observed (Finn et al. 2010, Robinson et al. 2014, Lencioni 2018). The newly-formed habitats following glacier recession are rapidly colonized by some alpine specialists. Alpine Chironomids (Chironomidae) constitute the most commonly observed taxon, but other taxa such as periphyton-grazing mayflies (Ephemeroptera) or detritus feeding stoneflies (Plecoptera) were also described. These species can colonize the streams which will be formed after glacier recession (Robinson et al. 2014). However, the lengthening of the streams is a transient phenomenon, concurrent to the recession of the glaciers. The disappearance of a glacier will eventually cause a significant impact on the biota colonizing these systems (Milner et al. 2009, Brown and Milner 2012): it could lead to a homogenization or a disappearance of these distinctive habitats, also due to the massive change in flow regimes when streams are no longer glacier-fed. Alterations to biotic interactions, food web structure, and ecosystem functioning can be expected (Robinson et al. 2014).

Aside from effects of climate change on macroinvertebrates in alpine streams, effects of climate change due to desiccation and intermittency may be most severe. Rüegg and Robinson (2004) examined the composition of the communities in streams with different intermittency regimes. They found that the density and richness were higher in permanent streams compared to temporary streams. Thus, increasing stream intermittency will likely have a negative effect on macroinvertebrate biodiversity. The period of the year when the drying occurs also affects the dominance of macroinvertebrate species in the communities, and will change biomass distribution. Organismal traits also differ between perennial and intermittent stream, with higher synchrony and faster growth in the latter.

Climate change is expected to increase the intensity of floods and to modify their seasonality toward earlier flood season. Such changes will likely affect macroinvertebrates, as bigger floods have detrimental effects on most taxa. The persistence of the taxa will strongly depend on their mobility, but even the presence of aerial adults cannot guarantee their persistence. Modifications of flood seasonality will alter emergence timing, as insects tend to emerge earlier to avoid predictable disturbances (Gray and Fisher 1981, Lytle 2002).

8.1.3 Macrophytes

Climate change has a counterintuitive effect on macrophytes: while higher temperatures and CO_2 concentrations are predicted to boost their growth rate and extend the growing season, they often react negatively to climate change due to interactions with phytoplankton. Such decreases in macrophyte populations have been reported in lakes across the globe (Zhang et al. 2017). As with all other organisms, climate change effects on macrophytes interact with multiple other factors: intensified human activities such as land use changes, aquaculture, and eutrophication also play major roles.

In addition to the general warming, the increase of extreme climatic events (both in frequency and intensity) can strongly impact macrophytes. Floods and storms limit light availability by increasing turbidity, and could cause the loss of both submerged and emergent vegetation (Valk 1994, Havens et al. 2004). During these events, conditions are more eutrophic due to transport of nutrient by precipitation, with potential consequences on phytoplankton growth, leading to competition for light and further reduction in macrophyte cover (Jeppesen et al. 2009, Short et al. 2016).

8.1.4 Diatoms

Diatoms are a very common unicellular microalgae group. They are important primary producers and indicators used for the reconstruction of past climate changes (Finsinger et al. 2014) and to evaluate water quality (FOEN 2007). As a consequence of climate change effects on lake physics (in particular stronger stratification), the seasonality of the diatom peak has changed in lakes (Yankova et al. 2017). The reduction of lake mixing lowers the concentration of P in the epilimnion. This phenomenon leads to higher silicon to phosphorus (Si:P) ratios, which is critical for diatoms as they have a silicon wall. As a consequence, dominance dynamics in the diatom community has changed. Until 2000, centric diatoms (Coscinodiscophyceae, cells with radial symmetry) were abundant in spring as they were adapted to low Si:P ratios, which are expected after the mixing of the P-rich deep waters and the nutrient-poor surface waters. In Lake Zürich, density of centric diatoms has recently declined, and their maximum density level has shifted toward summer (Yankova et al. 2017).

A second type of diatoms, pennate diatoms (Bacillariophyceae, cells with bilateral symmetry), is more adapted to high Si:P ratios and is therefore less affected by the change in nutrient concentrations and dynamics (Figure 8.4). These changes in diatom community structure and biodiversity due to climate change will affect their use as indicators of water quality (FOEN 2007, Visco et al. 2015), such that changes in their community due to climate change may be affecting the Diatom Index, which had been designed to assess freshwater ecosystems with respect to their nutrient content. Thus, it may be important to consider how the effect of climate change influences the interpretability of the Diatom Index.

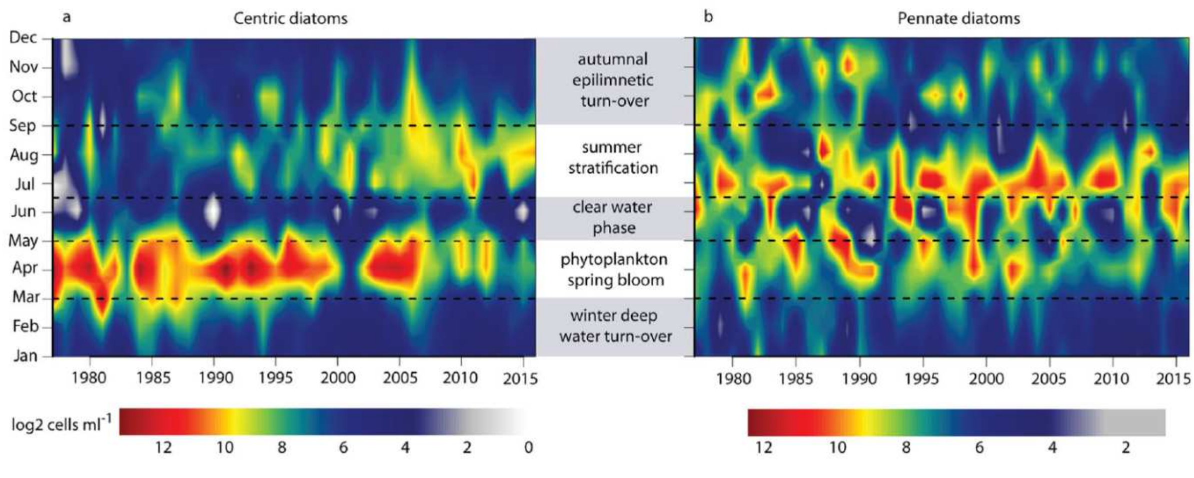


Figure 8.4: Long-term (1977 to 2016) trends in seasonal succession of (a) abundance of centric diatoms, (b) abundance of pennate diatoms (n = 480 for all parameters) during characteristic phenological phases (gray-white boxes). From Yankova et al. (2017).

8.1.5 Phytoplankton

Phytoplankton refers to all autotrophic types of plankton (i.e., small, mostly unicellular organisms, such as unicellular algae). To understand the changes in phytoplankton communities in Switzerland, it is important to take into account the history of the lakes, and specifically that the warming we now observe is concomitant with re-oligotrophication of the lakes (Monchamp et al. 2017). Eutrophication in Swiss lakes due to high nutrient loading had enormous effects on lake phytoplankton dynamics, and changed the community dominance from eukaryotic phytoplankton to cyanobacteria (Paerl and Huisman 2008, Sukenik et al. 2015). Specifically, smaller (phytoplanktonic) organisms such as cyanobacteria benefit even more from the warming, which then often leads to the recession of macrophytes through increased competition for light and nutrients (Short et al. 2016).

Monchamp et al. (2017) studied cyanobacterial community dynamics in 10 Swiss lakes over several decades, covering the eutrophication phase (~1960–1990) as well as subsequent re-oligotrophication and warming phases (remarkably strong since 1990). Overall, lake phytoplankton communities were strongly impacted by eutrophication, re-oligotrophication, and climate change. The primary effect of climate change on phytoplankton arises from stronger stratification, associated with increased water column stability and a strengthened gradient of environmental conditions (e.g., nutrients are more abundant in the deeper water layers) (Klausmeier and Litchman 2001, Stomp et al. 2004, Longhi and Beisner 2009). This gradient favors cyanobacteria that can regulate their buoyancy and thus access both light at the surface and nutrients under the photic zone (Posch et al. 2012, Monchamp et al. 2017). Among cyanobacteria, *Dolichospermum lemmermannii* (Nostocales), *Planktothrix rubescens* (Oscillatoriales), and the genus *Microcystis* (Chroococcales) have particularly increased prevalence and geographic range over time across lakes in Switzerland (Figure 8.5). These taxa comprise bloom-forming and toxic organisms.

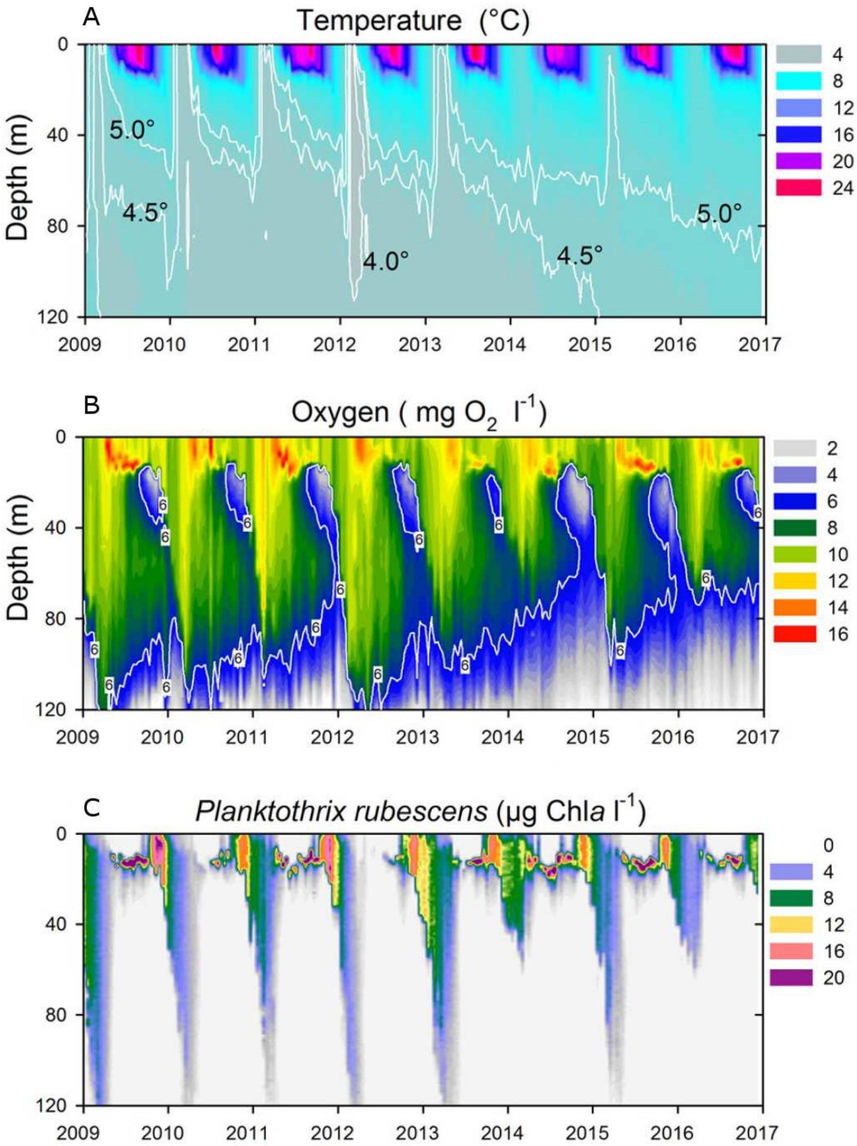


Figure 8.5: Recent trends (2009–2016) in warming, concentrations of oxygen and cyanobacterial biomass in Lake Zürich (Switzerland). A) Water temperature and two white isolines showing 4.5 °C and 5.0 °C, respectively. B) Dissolved oxygen concentration and the 6 mg O2 l^{-1} isoline, which is a proxy for the depth of maximal water turnover during spring. Note the metalimnetic oxygen minima developing each autumn expanding between 15–40 m water depths. C) Total biomass (chlorophyll a concentration) of the most dominant primary producer in Lake Zürich, the cyanobacterium *Planktothrix rubescens*. Data based on biweekly profiles (n = 192) of parameters measured in 1 m depth intervals (0 to 120 m depth). Adapted from Yankova et al. (2017).

Cyanobacteria richness is increasing in Swiss lakes; however, community similarity across lakes is also increasing (Figure 8.6; Monchamp et al. 2017). Within lakes, environmental gradients shaping depth structure also influence the assemblage. With warming and reduced nutrients, communities are for example more clustered in summer and autumn in Lake Zürich (Pomati et al. 2017). Yankova et al. (2017) found no evidence of cascading effects onto diatoms' consumers.

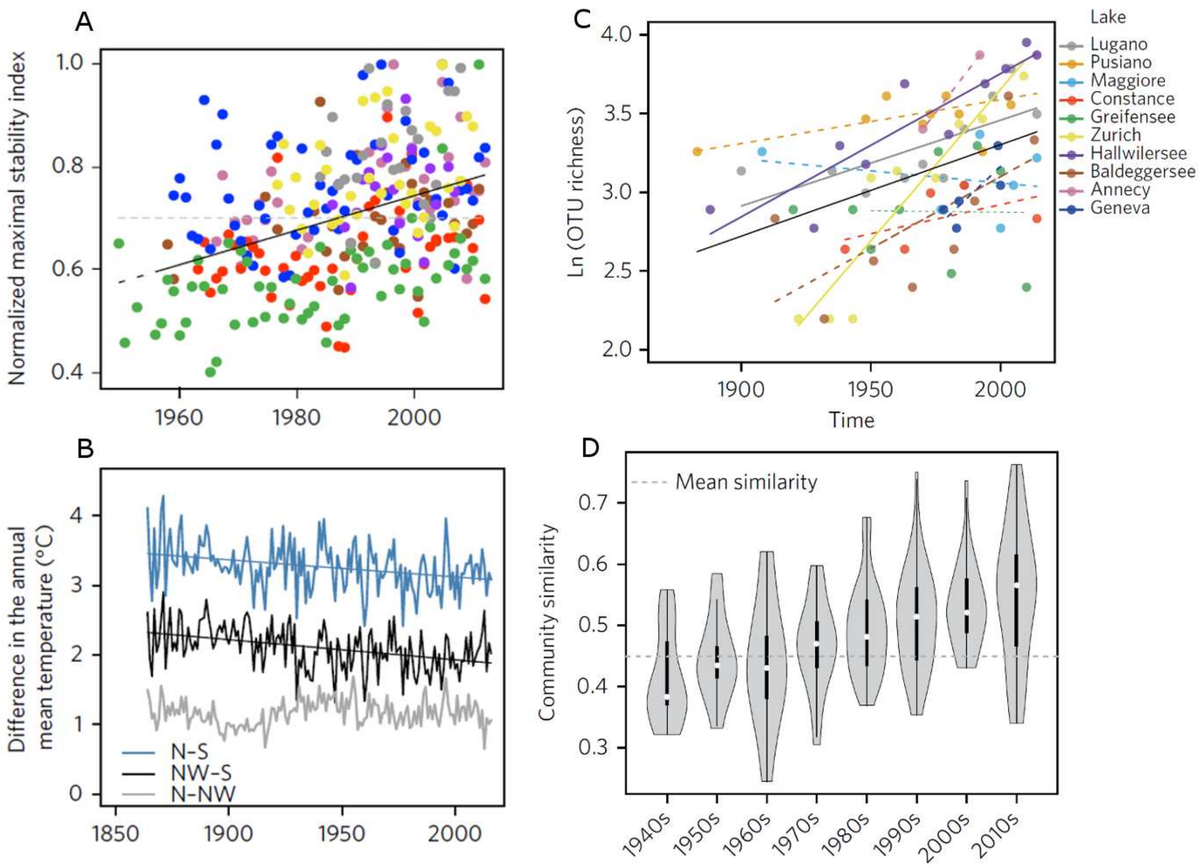


Figure 8.6: History of environmental conditions and community richness and similarity in ten perialpine lakes. A) Normalized stability index (SSI – stability of the water column) values plotted against time. The dashed line shows the average SSI value over all lakes and time points, and the black line shows the significant linear fit. B) Pairwise difference between the mean annual air temperatures at three meteorological stations. N: Alpine north side–eastern Plateau; NW: Alpine north side–western Plateau; S: Alpine south side. Significant linear fits in two of the time series are indicated by a solid line. C) Temporal plot of natural log-transformed rarefied cyanobacteria OTU (operational taxonomic unit) richness. Colored lines show the lake-specific significant (solid lines) or non-significant (dashed lines) relationships and the black line shows the overall fit for all lakes combined. D) Violin plots showing the probability density, median and interquartile range of pairwise unweighted Unifrac similarities estimated across lake communities for each decade. The dashed horizontal line represents the mean value of pairwise similarity across all samples. Adapted from Monchamp et al. (2017).

Cyanobacteria blooms are often detrimental to other species because they alter environmental conditions. Blooms reduce the light levels available for other phytoplankton species, and impact higher trophic level organisms as zooplankton and fish (e.g., by causing anoxia) (Vonlanthen et al. 2012, Sukenik et al. 2015). The toxicity of the bacteria is also a concern as the bloom can contain harmful cyanobacteria, which seems to be a growing phenomenon worldwide (Sinha et al. 2012, Salmaso et al. 2015, Salmaso et al. 2017). This may be one of the most obvious and direct impacts of climate change on aquatic ecosystems, and will become directly relevant and obvious to the public.

8.2 Genetic diversity

In the past, most research into the effects of environmental change on ecological systems has focused on changes in species richness/composition. Recently, it has become clear that effects on genetic diversity may also be highly relevant (e.g., Paz-Vinas et al. 2015; Fourtune et al. 2016; Seymour et al. 2016a, 2016b), especially in the context of (potential) future genetic adaptability to novel environments, such as during climate warming. Hotaling et al. (2017) reviewed current knowledge on population genetics of organisms in alpine streams and concluded that despite the valuable information it represents to conservation biology and to understand evolutionary processes, the subject has been hitherto mostly overlooked. The authors concluded that next generation sequencing could provide this information, but has been underused in alpine stream biology.

In mountainous environments, commonly found in Swiss freshwater systems, the high degree of isolation results in limited gene flow, and the amount of genetic variation is therefore relatively high (Finn et al. 2006, Hotaling et al. 2017). The decrease in habitat heterogeneity associated with glacier shrinking (Brown et al. 2007) and the homogenization of lake temperatures and mixing dynamics (Monchamp et al. 2017, Yankova et al. 2017) could lead to a loss of genetic diversity (Hotaling et al. 2017). Genetic diversity loss is predicted to be higher than the expected loss when using morphologically defined species (Hotaling et al. 2017). In a study led by Bálint et al. (2011) and focusing on the impact of climate change on population genetics, 67 % of named species considered in the analysis were predicted to persist under two CO_2 emission scenarios, while only 35 to 16 % of diversity persisted when using genetic metrics (Figure 8.7). This loss in genetic diversity can have strong implications as it may reduce phenotypic plasticity and local adaptation. As a consequence, populations could lose the potential to adapt to changing environment, including further changes resulting from climate change (Hughes et al. 2008, Bálint et al. 2011).

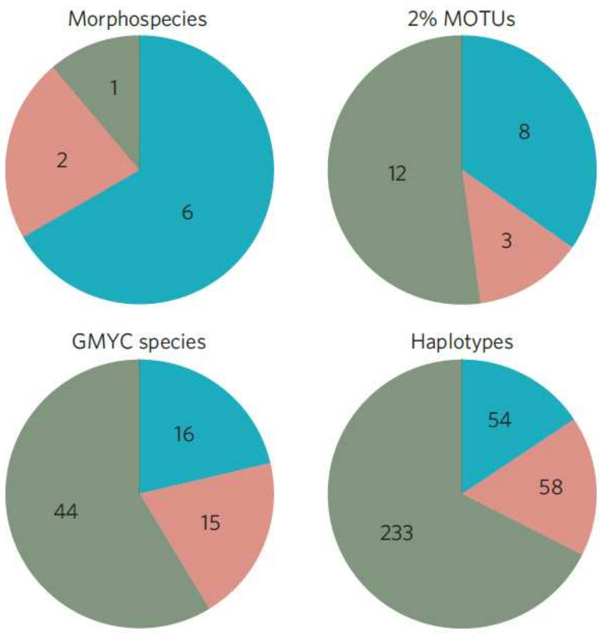


Figure 8.7: Projected losses of morphologically identified species, molecular operational taxonomic unit (MOTUs), general mixed Yule-coalescent (GMYC) and mitochondrial DNA haplotypes combined for nine montane aquatic insect species in Europe under two IPCC 2080 CO_2 emission scenarios. Absolute numbers of units that will persist under both future climate projections (blue), units that will be lost only under the 'business as usual' scenario (A2a; pink) and units lost under both scenarios (grey). MOTUs, molecular operational taxonomic units. From Bálint et al. (2011).

Finn et al. (2013) studied whether and how (partial) glaciation of catchments affected genetic diversity of macroinvertebrates compared to catchments without any ice coverage. Beta-diversity was the highest in highly glaciated reaches, and diversity turnover was the greatest between high- and mid-glacial reaches. These trends are probably driven by the particular conditions in these environments: low temperature, high instability and isolation. Glacial reduction is therefore likely to reduce genetic diversity.

9 Effects on life history

Some of the most pronounced effects of climate change on organisms are changes in their life history and phenology, such as changes in voltinism[5], changes in flight period, and seasonal appearance (Altermatt 2010a, Altermatt 2010b, Walther 2010). Most of the research on this has been conducted in terrestrial systems, with a focus on plants, insects and birds (Walther et al. 2002). Studies in aquatic systems, however, are generally in accordance: organisms change their phenology and hatch earlier, but also often have lower survival due to the increased temperatures and the subsequent increase in metabolic rates, such as is documented for dragonflies or mayflies (e.g., Dingemanse and Kalkman 2008, Everall et al. 2015, Mccauley et al. 2015). These changes in phenology have similar consequences as in terrestrial systems: species interactions may be disrupted, for example between fish and their aquatic insect prey, or the timing of emergence of aquatic insects may change and no longer match phenology of their terrestrial predators. Furthermore, changes in phenology may affect the protocols of monitoring programs. Common programs, such as the BDM or NAWA, have specific time windows defined to collect aquatic macroinvertebrates at different elevations. Changes in phenology, such as an advanced emergence, may shift the organisms outside these time windows optimal for detection, causing false-negatives. Thus, in the mid to long term, these time windows need to be adjusted.

The most pronounced life history changes are expected in alpine systems, because that is where phenological windows are often most constrained. Life histories of alpine aquatic species, however, are not well understood, even though species' persistence depends on particular traits (e.g., development rate, emergence timing, size at maturity, reproductive traits and other traits allowing resistance or resilience) (Hotaling et al. 2017). Most of the studies focusing on this topic considered macroinvertebrates. Due to the particular conditions in alpine regions (e.g., strong temperature gradients, disturbance regimes), most aquatic insect species show rapid responses for a large number of life history traits. These changes in traits can be plastic[6], adaptive[7], or both (Lytle and Poff 2004, Poff et al. 2006, Lytle et al. 2008, Hotaling et al. 2017). This leads to differences observed in the field, where length of the life cycle, body size, number of eggs laid vary with altitude, and patterns that can differ from one species to another (Shama and Robinson 2009, Hotaling et al. 2017).

9.1 Phenology

At global scale, climate change is affecting the phenology of the species, with ectothermic species being most strongly affected (Cohen et al. 2018). This study shows that overall

[5] Number of generations of an organism in a year

[6] Phenotypic plasticity refers to the changes as a response to the environment

[7] Adaptive traits are heritable

phenology (e.g., breeding time, migration) occurs earlier. Such changes in phenology can lead to mismatches within food chains (Woodward et al. 2010b, Cohen et al. 2018) subsequently affecting communities and ecosystems.

Larger quantities of snow delay the emergence of insects such as caddisflies, stoneflies, and mayflies (Finn and Poff 2008, Shama and Robinson 2009). The presence of glaciers is also important for phenology, with individuals close to glaciers reaching pupation earlier in the season (Shama and Robinson 2009). Overall, the most commonly observed changes in phenology are advances of emergence and more rapid development.

9.2 Physiology and species traits

Physiology of most aquatic organisms is tightly linked to water temperature. Thus, climate change may have very direct effects on physiology. Counterintuitively, the response to temperature change in alpine regions could be less dramatic than at lower altitudes, because ectotherms at higher altitudes (and latitudes) have broader thermal ranges (Woodward et al. 2010b, Hotaling et al. 2017). However, in alpine regions the capacity of some species to resist to persistent cold and freezing trades off with their capacity to persist in warmer habitats (Lencioni and Bernabò 2017). These species could therefore be outcompeted by generalist as the environmental conditions change toward warmer temperatures. This phenomenon was observed for the midge genus *Diamesa* (Chironomidae). Flory and Milner (2008) proposed that this might represent a general pattern in cold-adapted macroinvertebrate communities.

Extreme events such as heat waves or heavy precipitation threaten persistence of species, and possessing certain traits can determine the ability of organisms to overcome these disturbances (Poff et al. 2018). At the larval stage, mobility is the most important trait to face flooding from heavy precipitation. Taxa that are able to swim or drift can persist under high disturbance levels, while low mobility taxa that crawl cannot. Yet among these low-mobility taxa, crush-resistant species were an exception, and were able to persist more even though they were strongly impacted (Poff et al. 2018).

Trophic traits can also shift toward allochthonous[8] food sources as a result of changes in the environment (e.g., new prey, land use, or vegetation shift from evergreen to deciduous). Specialist species losing their food source are less likely to survive such changes, as trophic traits tend to be phylogenetically constrained (Poff et al. 2006). Therefore, assemblages will be modified, with potentially strong changes in functions and ecosystem-level processes (Robinson and Gessner 2000, Cauvy-Fraunié et al. 2016).

10 Effects on invasion dynamics

Invasions of non-native species are among the most severe causes of biodiversity decline and altered ecosystem dynamics in aquatic ecosystems (Leuven et al. 2009). While most invasions are associated with anthropogenic activities, climate change further propagates the success of non-native species in aquatic systems (Rahel and Olden 2008). Climate change is expected to affect all stages of biological invasions (Figure 10.1; Walther et al. 2009).

---

[8] materials imported into an ecosystem

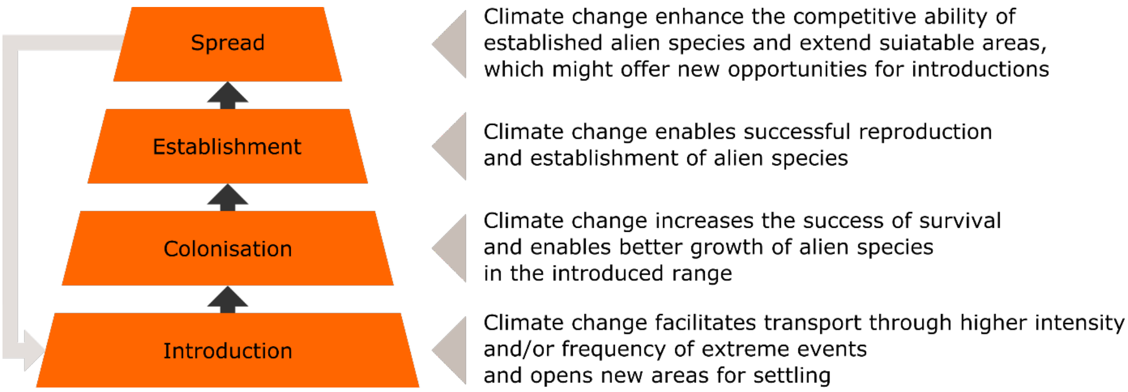


Figure 10.1: Influence of climate change on all the sequential transitions of a successful invasion process. Adapted from Walther et al. (2009).

In a comprehensive review, Rahel and Olden (2008) identify that climate change will influence the probability of new non-native species to establish, for example by eliminating environmental filters such as cold temperatures or winter hypoxia in lakes. These two factors currently prevent survival of many non-native species in aquatic systems. Furthermore, the construction of reservoirs to mitigate runoff changes due to climate change may increase invasion success, as these reservoirs serve as hotspots for invasive species. The connection of distant basins for human water needs could also increase the distribution of invasive species (Galil et al. 2008, Walther et al. 2009). Climate change will also modify the ecological impacts of invasive species by enhancing their competitive and predatory effects on native species, and by increasing the virulence of some diseases. A direct result of climate change may be the need for novel strategies to prevent and control invasive species, such as barrier construction. At the early stage of an invasion (introduction), climate change may help invasive species by removing physical constraints like overwintering in temperate climate or by changing the phenology or dispersal ability of the species. These effects have been observed for the establishment of non-native aquatic organisms (in warm and dry years), and facilitate the introduction of new species (Winder et al. 2011).

After colonization, a species must be able to reproduce if it is to persist. If warming increases the rate of reproduction, it will favor the establishment of alien species. This is especially true for species coming from warmer regions. Longer growing seasons also lead to a larger number of generations, and bigger populations with longer reproductive periods. Climate is important for population abundance and distribution. By increasing reproduction and population size, it also promotes subsequent invasions from the newly colonized area. However, the changes do not always favor the alien species, and changes can also increase the potential of the native species (Walther et al. 2009). Climate change should favor species that are able to tolerate warmer and more variable climatic conditions, resulting in a relative increase in their performance and/or movement to new locations.

Understanding the effect of climate change on invasions is important, because invasions can strongly impact ecosystems (Gallardo et al. 2016). This can be through direct impacts on ecosystems, for example competition or predation, or through indirect effects, such as changes in habitat properties or cascading effects through trophic networks. Invasions in freshwater generally decrease the abundance and diversity of native communities (Gallardo et al. 2016). A large evidence base indicates that in invaded habitats, the abundances of macrophytes, fish, and zooplankton often strongly decrease, while there are no clear trends for benthic invertebrates and phytoplankton. Most of the work on the effect of climate change on invasion dynamics does not take into account changes in precipitation or runoff regimes, as they are more difficult to predict (Walther et al. 2009). Such drivers, however, are expected to affect invasion dynamics by, for example, modifying connectivity.

A few key lessons on climate change effects on invasion dynamics are:

- climate change will affect aquatic invasive species throughout the invasion pathway (Hellmann et al. 2008);
- most activities to mitigate or control non-native species do not (yet) take climate change effects into account, potentially jeopardizing management goals (Bellard et al. 2013);
- capacity and knowledge exists to incorporate this information, although tools may be lacking (Rahel et al. 2008);
- there are important research gaps in understanding how climate change affects and interacts with other stressors (Walther et al. 2009, Jourdan et al. 2018); and
- more information is crucially needed to better understand impacts and adaptation of non-native species for effective management (Driscoll et al. 2012).

11 Effects on ecosystem function

Dynamics at the ecosystem level in freshwater systems are very complex, including input, transformation, and transport of organic matter, and there are many interactions among these dynamics and processes (e.g., Gounand et al. 2018, Little and Altermatt 2018). We showed in the preceding chapters that changes resulting from climate change are very diverse, arise from different components of climate change, and comprise both direct and indirect effects (e.g., temperature, precipitation, lake stratification). Importantly, the effect of climate change at the ecosystem level is not the sum of the effect on the lower levels of organization. Indeed, many ecosystems properties emerge from complex interactions (Petchey et al. 2004, Walther 2010, Woodward et al. 2010b).

To address the question of the effect of climate change at the ecosystem level, and to disentangle the effect of climate change from other global or local changes, most studies consider very simplified or specific ecosystems. These studies are valuable, but do not allow to conclude or extrapolate for predictions at the ecosystem level in the context of the climate change that Switzerland is and will be facing. Even when trends are identified concerning effects of climate change at the ecosystem level, many of the underlying phenomena are not mechanistically understood.

11.1 Effects on abundance/productivity

Primary productivity is, together with allochthonous inputs, the source of energy of all ecosystems. Changes in primary production can lead to strong shifts in communities (e.g., reduction of biomass or reorganization of food webs). Most metabolic rates increase with temperature, which is subsequently expected to boost primary production (Friberg et al. 2009, Woodward et al. 2010b, Rasconi et al. 2015, Yvon-Durocher et al. 2015). However, as respiration also increases, and as most freshwater lotic ecosystems depend primarily on allochthonous inputs, the relevance of these changes is difficult to assess.

In order to test the hypothesis of increased primary production due to climate change in aquatic systems, experiments where conducted in mesocosms and streams that were warmed to simulate the effect of climate change. Primary production increased (Rasconi et al. 2015), as did net C production (Schmid 2014). However, this does not seem to be a universal pattern, and alternative outcomes have been observed as well, highlighting the contrasting effect of climate change on aquatic ecosystems. With warmer temperatures, respiration rates will also increase (Kritzberg et al. 2014, Atwood et al. 2015, Yvon-Durocher et al. 2017). Measuring gross primary production (GPP) is therefore important, as primary producers increase their respiration and modify the fluxes. Respiration has been found to be more related to temperature than GPP, and therefore increases faster (Demars et al. 2011).

In a study led by Garnier et al. (2017), the effects of four environmental disturbances (temperature, nutrient ratio, C enrichment, and light) on aquatic communities were tested in a full-factorial experiment, measuring biomass and dissolved oxygen as ecosystem functions. The concentration of dissolved oxygen indicates the relative importance of respiration and photosynthesis (a decrease in dissolved oxygen relates to an increase in respiration). Garnier et al. (2017) observed that C enrichment had the largest effect on dissolved oxygen, strongly decreasing its concentration. They also observed that the community was resilient and recovered from the perturbation. Temperature alone did not significantly affect the recovery of the system. However, when temperature was variable, it reduced resistance and increased the return time to a comparable concentration of dissolved oxygen. In contrast, Liboriussen et al. (2011) observed modifications only for the sediment respiration and higher respiration only occurred when nutrient levels were kept constant. These results reinforce the importance of taking into account the other aspects of climate change and the potential effect on nutrient levels – for example, nutrient increases and temperature increases can counterbalance each other and lead to no net changes (Liboriussen et al. 2011). Not only absolute changes in total primary production are expected, but also the seasonality of primary production is expected to shift and the peak of the productivity will be reached earlier in the year (Yvon-Durocher et al. 2017).

Changes in productivity and fluxes also influence emissions at the ecosystem level. Yvon-Durocher et al. (2011b, 2017) observed increased emissions of CO_2 and methane in mesocosms set under warmer conditions, suggesting that ponds could become net producers of greenhouse gases. Such feedbacks are hard to anticipate, but could have large consequences, as CO_2 emissions could be doubled in these systems (Boyero et al. 2011, Demars et al. 2011, Atwood et al. 2015).

As a general biological rule, organisms' biomass decreases with temperature (Figure 11.1; Brown et al. 2004, Nelson et al. 2017a), suggesting that climate change related warming could also change the size-structure of aquatic communities. Some studies support this hypothesis (Yvon-Durocher et al. 2011a), while in other studies biomass is stable (Özen et al. 2013, Schwarzer et al. 2013, Urrutia-Cordero et al. 2017) or increases (Nelson et al. 2017a). Again, these changes often seem to be specific to the study community, and both thermal physiological traits from the regional species pool and the energy constraints are important factors to take into account (Nelson et al. 2017b).

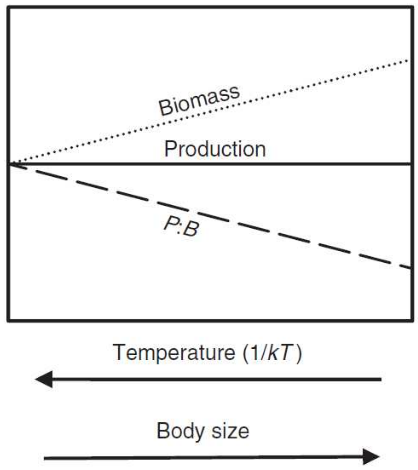


Figure 11.1: Predicted relationships between temperature, body size, biomass, production and biomass turnover (P:B) derived from Brown et al. (2004). From Nelson et al. (2017b).

11.2 Effects on resource turnover (decomposition)

In freshwater ecosystems, decomposition is an essential process, as allochthonous resources from terrestrial ecosystem represent a substantial food web flux (Collins et al. 2015). Climate change will increase litter input from riparian vegetation, increasing the amount of organic matter available (Wrona et al. 2006). This change is likely to strongly affect oligotrophic alpine lakes located below the treeline. Leaf litter breakdown, which is essential for the decomposition of allochthonous resources is stimulated by temperature, could lead to higher levels of nutrients in the streams (Friberg et al. 2009). Decomposition rate, and thus accessibility and integration of these resources into the aquatic ecosystem, is expected to increase with increasing temperature, following the same trend as other metabolic rates (Woodward et al. 2010b). Nutrient cycling is also predicted to be faster (Demars et al. 2011). However, experimental work on this topic showed complex responses and sometimes counterintuitive or even contradictory results. In a study led by Dossena et al. (2012), decomposition increased with increasing temperature. This change was driven by a shift in community composition toward bigger invertebrate taxa and an increase of the microbial activity in fall. In another experiment, decomposer communities changed and the microbial community contributed more to decomposition. However, the invertebrate communities were reduced and this shift toward microbial-dominated decomposition led no net change in decomposition rate (Boyero et al. 2011). Yet another study suggests that the change will be seasonal, and that decomposition will only increase in winter, as temperatures will rise to levels allowing microbial activity (Ferreira and Chauvet 2011).

To conclude on this topic, seasonality seems to be critical in multiple aspects. Furthermore, measuring fluxes at the scale of the different trophic compartments is important, as strong shifts between microbes and invertebrates have been observed.

11.3 Effects on other ecosystem properties

11.3.1 Effect on food webs

The metabolic theory of ecology predicts that biomass of top predators should decline with warming (Brown et al. 2004, Woodward et al. 2010b). However, several experiments observed top-heavy food webs, indicating strong top-down control (Shurin et al. 2012, Atwood et al. 2015, O'Gorman et al. 2016) linked with an increase of activity, foraging, digestion, growth rate of predators (Hoekman 2010), and metabolic demand of predators (O'Gorman et al. 2017). These changes are also linked to the nutrient availability, because high resource levels are required to sustain these top-heavy food webs.

These results were obtained from warming experiments, and more investigation is needed to better understand the role that predator and prey identity, food chain length, and diversity play in natural systems (Atwood et al. 2015). Water temperature also affects communities indirectly, by changing development rates and subsequently affecting food webs through changes in phenology, causing mismatches within the trophic chain (Cohen et al. 2018).

Heat waves leading to fish kills (top predators) can have cascading effects on the food web lasting for several years, especially if recolonization of the area is slow due to low connectivity or isolation (Wrona et al. 2006).

Changing flow regimes will also affect food web structure. A modelling approach predicted that there would be more trophic levels (increased food chain length) in rivers with more variable flow (as expected in the context of climate change) than in rivers with stable flow.

11.3.2 Effect on the size structure of the communities

Theory predicts that warming will favor small organisms, as large individuals are more likely to disappear as an result of reduced abundance (Brown et al. 2004, Woodward et al. 2010b). Experimental warming experiments have generally shown this pattern (Petchey et al. 1999, Yvon-Durocher et al. 2011a, Rasconi et al. 2015, O'Gorman et al. 2017). However, for some specific organisms, like diatoms, no such changes were observed (Adams et al. 2013), and some studies showed opposing effects (Yvon-Durocher et al. 2015, Nelson et al. 2017a). These unexpected changes were attributed to reorganization of the ecosystem such that the energy balance was maintained and a new community was formed from the regional species pool. Another experiment showed that changes can be seasonal, with a steeper size spectrum in the warmed communities (Dossena et al. 2012).

Our understanding of the effects of climate change on aquatic ecosystem properties is still limited and despite efforts to date, no consensus on the subject has yet been formulated. It is not possible to predict effects in this domain for Switzerland with any confidence.

12 Indicators and assessment methods/monitoring methods

Various ecological monitoring and biodiversity assessments are already implemented in Swiss freshwater systems (e.g., BDM, NAWA etc.). However, to trace the effects of climate change, some additional measures are critical. For example, monitoring genetic and phenotypic diversity is highly relevant, as the earliest diversity losses can only be observed at this level (Bálint et al. 2011, Brodersen and Seehausen 2014). Furthermore, most monitoring focuses on rivers and streams, and little extensive monitoring is implemented in lakes, although many effects of climate change in aquatic systems are expected to occur in lakes and standing waters.

There are two types of considerations to make in the context of monitoring/assessment of aquatic ecosystems in a changing climate. First, it is essential to understand whether and how indices of currently-implemented monitoring programs, for example those focusing on water quality or the ecomorphology of rivers (FOEN 2013, BDM Coordination Office 2014), will be affected by climate change, and to address any possible effect on the indices which would change the conclusions. For example, biomonitoring indices have been reported to be dependent on stream intermittency, which will be affected by climate change (Wilding et al. 2018). As another example, when climate change results in a change in the community composition, this could affect the index such that it is interpreted as a change in water quality, while the latter actually did not change. Thus, the robustness of any monitoring programs and assessments, and their respective indices, to possible climate change effects must be evaluated. Second, the effect of climate change on freshwater systems itself needs to be monitored, and most current monitoring does not include that as a specific target. Either these monitoring efforts need to be complemented/adapted, such that their data can also be used to assess climate change effects on aquatic ecosystems, or new monitoring programs should be established. This also highlights that monitoring programs and assessments should be both specific enough to address individual environmental changes/stressors, but also be generic, such that new or different stressors can be equally well captured. The current monitoring strategy may not be ideal in this regard.

In addition to traditional biodiversity monitoring, taking functional diversity into account in the monitoring design could also elicit more information, especially in context of climate change, and help conservation of biodiversity. Information on thermal traits and dispersal are crucial to evaluate which species are the most threatened by climate change (Conti et al. 2014, Hershkovitz et al. 2015). Sievert et al. (2016) proposed an indicator of the vulnerability of fish in the context of climate change, using information about the habitat preference, tolerance to flow, and temperature changes, and combining this with dispersal and range data, rarity, and habitat resilience. Tolerance to flow and temperature change was assessed using two different approaches: traits and documented species responses. The second approach has a clear limitation because the conditions expected after climate change were only experienced by the species during an extreme event, and may not capture future states of the streams and the effect on stream fish communities. This type of index can nevertheless provide a framework to redefine the lists of endangered species including the risk associated with climate change. In another study, Hershkovitz et al. (2015) used a trait-based approach and included distribution data. The factors considered were endemism (species that are only found in one of the 25 European ecoregions), micro-endemism (endemic species that are found only in a specific geographical area within an ecoregion), headwater preference (species restricted to upper reaches), high altitude preference (species preferring streams at elevations > 800 m), cold water preference (species that only exist in low water temperatures < 10 °C), long-lived species (species with a life span > 1 year), uni-voltine and semi-voltine species (species that only reproduce every year and every 2 years, respectively). According to this evaluation, the alpine region was the zone where the highest number of vulnerable species were located. Such methods allow a focusing of conservation efforts on species that are particularly sensitive to climate change, and complete the current methods of risk assessment for protected species and area.

In conclusion, monitoring the effect of climate change is a challenge, because multiple stressors have effects on the aquatic ecosystems. Monitoring methods should integrate variables that allow one to disentangle the effect of these stressors and implement targeted mitigation actions. A revision of the monitoring methods in the light of the effect of climate change is necessary, and monitoring methods and conservation actions should integrate more genetic and phenotypic diversity in their evaluations.

13 Knowledge gaps

First of all, climatic predictions remain unclear, especially concerning **hydrological changes**. This was already observed by Bates et al. (2008), who stated that “the ability to quantify future changes in hydrological variables, and their impacts on systems and sectors, is limited by uncertainty at all stages of the assessment process. Uncertainty comes from the range of socio-economic development scenarios, the range of climate model projections for a given scenario, the downscaling of climate effects to local/regional scales, impacts assessments, and feedbacks from adaptation and mitigation activities”. These authors identified research gaps in monitoring, climate modeling, the impact of extreme events on water quality, and adaptation/mitigation strategies. Most of these gaps still exist.

Lack of knowledge about hydrological processes makes it difficult to form accurate predictions for related processes:

- Many impacts of altered precipitation regimes are still under active research. As a few examples, it remains partly unclear (i) how the future flow regime of rivers will impact water chemistry and quality (U. von Gunten, personal communication), (ii) to what extent strong warming of urban streams following summer storms affects the aquatic ecosystem (P. Niederhauser, personal communication), (iii) how river intrusions will contribute to water and oxygen renewal in deep lakes, and (iv) what impact future summer droughts and storms will have on karst groundwater quality (U. von Gunten, personal communication).
- Snowfall in future winters is still poorly predicted by climatic models (F. Hagedorn, personal communication). More reliable predictions would allow better quantification of effects related to snow cover, such as higher microbial activity in alpine, snow-covered environments, where soils are insulated by snow and remain warmer.

Concerning **soils and catchments**, physicochemical processes and their response to temperature and humidity changes are generally well known. Still, predictions for the future are difficult because (i) high-resolution spatial field data on soil characteristics are lacking, so that knowledge about catchment conditions is incomplete (E. Frossard, personal communication), and (ii) change is often determined by the cumulative effect of several “small” drivers, which can act in the same or opposing directions (V. Prasuhn and W. Richner, personal communication). As such, the overall trajectory of soil condition remains unclear (U. von Gunten, personal communication). Also, interactions between mechanisms acting in the hyporheic zone and the effects of external drivers are still poorly understood, so that the significance of this zone to stream ecology and biogeochemistry is still a major research topic (Boano et al. 2014).

The feedbacks between climate change and land use (e.g., forestry and agricultural practices) remain underresearched (Bates et al. 2008). Vegetation change, for example, is complex to predict, but can have large effects, which are not well understood. It can for example affect nutrient cycling in soils and waterbodies, wind protection, shading, and bank stability. Other changes, such as tree community composition, will result from the interaction between climate change and land use. Community shifts could lead to changes in resource inputs to aquatic ecosystems, and alter decomposer communities (Little and Altermatt 2018).

Concerning the effects of climate change on **ecological dynamics**, knowledge gaps are mostly related to the effects of stream intermittency and tipping points (resilience), to the life histories of alpine aquatic species, and to evolutionary dynamics. For example, life histories of many aquatic organisms, especially invertebrates, are not well understood in the context of climate change, yet persistence of these species depends on particular traits (e.g., development rate, emergence timing, size at maturity, reproductive traits and other traits allowing resistance or resilience) (Hotaling et al. 2017).

Furthermore, knowledge about climate change effects is distributed unevenly among species. While effects on some groups (e.g., fish) may be relatively well understood and predictable, other groups are less studied. For example, we lack much data about climate change effects on invasions of microorganisms (Amalfitano et al. 2015) in aquatic systems.

Finally, a major gap is the understanding of climate change on ecosystem processes. Even though some trends have been identified concerning effects of climate change at the ecosystem level, many of the underlying phenomena and mechanisms are not understood.

The impacts of climate change on water quality and ecological status in water-poor and/or developing countries (i.e., not Switzerland) have received little attention, although they will be of critical importance in the next decades. For instance, Bates et al. (2008) note the necessity of investigating the impact of wastewater discharges in developing countries.

14 Concluding remarks

In their Fifth Assessment Report, the IPCC writes: “the interaction of (i) increased temperature, (ii) increased sediment, nutrient and pollutant loadings from heavy rainfall, (iii) increased concentrations of pollutants during droughts, and (iv) disruption of treatment facilities during floods will reduce raw water quality and pose risks to drinking water quality” (IPCC 2014).

In Switzerland, this statement can be made more specific. Droughts and storms in summer affect transformation and transport of chemicals (typically, during and after rain), especially in urban and agricultural areas. However, over the whole summer, overall chemical loading is likely to be reduced, while in winter, it is expected that soils will be warmer and wetter (especially in the lowlands), favoring biochemical activity and increasing compound mobility. All in all, there are only few impacts on water quality (Figure 14.1) that cannot be prevented through management and adaptation. These include, for example, increasing water temperatures and the seasonal shift of the rivers discharge regime, implying decreasing flows in summer and autumn in non-regulated rivers. Critical impacts to lakes, such as decreasing near-bottom oxygen concentrations and cyanobacterial blooms in late summer, can be partially offset by better nutrient management in their catchments. Similarly, pollution peaks (e.g., CSOs in urban catchments, or plant protection products in agricultural catchments) can be prevented by promoting reduction of the inputs, and by better runoff and wastewater management. In Switzerland, much has already been done in order to protect our waters and it is to expect that (i) these past and current efforts will allow the retention of good water quality despite climate change, and (ii) further efforts, while useful and necessary, will be increasingly expensive to put into place. The biological impacts of climate change, however, are much less likely to be prevented or mitigated (Figure 14.2; Box 8), such that many of the effects of climate change on the ecological status of aquatic systems discussed in this report will probably happen.

Figures 14.1 and 14.2 summarize the main effects that are expected to occur as direct impacts of warming, and that affect both environmental conditions and the organisms living in freshwater ecosystems in Switzerland. The figure is based on the above-reviewed literature and can be seen as a visual summary of our work.

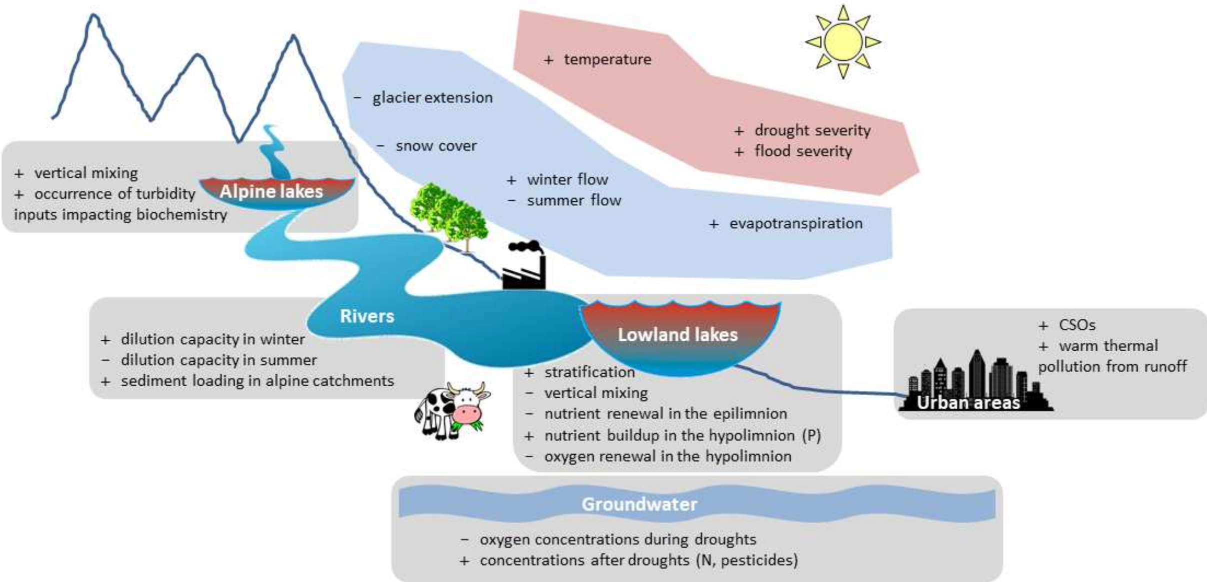


Figure 14.1: Visual synthesis of the impacts of climate change relevant to water quality.
Plus signs (+) indicate increases of a phenomenon, minus signs (-) indicate decreases of a phenomenon.

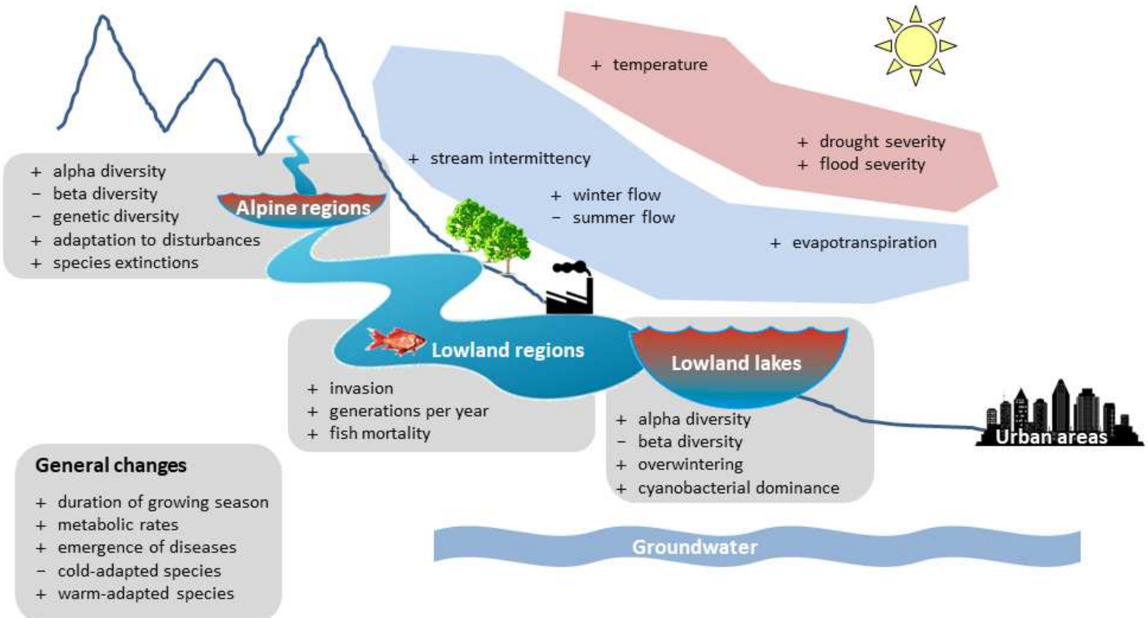


Figure 14.2: Visual synthesis of the impacts of climate change relevant to aquatic ecosystems.
Plus signs (+) indicate increases of a phenomenon, minus signs (-) indicate decreases of a phenomenon.

Box 8: Potential tipping points.

Although climate change causes slow gradual changes in environmental conditions, these changes may exceed some physical, chemical, or ecological thresholds and abruptly have a strong impact on the dynamics of waterbodies. Here, we give a few examples of this phenomenon, which is commonly referred to as a "tipping point".

Many of the large Swiss lakes are oligomictic, with full mixing occurring only during cold winters. Cold winters will be much less frequent with climate change, which will greatly reduce (or even interrupt) the occurrence of deep mixing. Such change could make the hypolimnion of lakes anoxic, and modify the biochemistry of the whole water column. At the lake surface, warmer conditions could induce a shift in the algal community towards cyanobacterial dominance during late summer. This could directly affect water quality during and after algal blooms, and impact the ecological balance throughout the year. With increased severity of summer droughts, larger rivers are at risk of drying. For a river that normally flows year-round, drying has dramatic ecological consequences: destruction of many habitats, death of eggs and organisms, and loss of migratory corridors. Long and severe droughts may also result, locally and temporarily, in oxygen depletion in sensitive aquifers. This implies a deterioration of water quality of valuable drinking water sources.

Biological tipping points (also called regime shifts) are often linked to abrupt changes in the dominance of single organisms, such as switches between clear and turbid water states, which are regularly found in shallow lakes. Such shifts are caused by positive feedbacks between water clarity and macrophytes (Scheffer et al. 2001, Scheffer 2004, Hilt et al. 2011). In such systems, relatively small changes in temperature or nutrient levels can drastically affect the abundance and occurrence of organisms, and switch lakes between clear water lakes dominated by macrophytes and turbid water lakes dominated by phytoplankton (Scheffer et al. 1993). Similar tipping points are found between permanent and intermittent streams. Streams shifting to intermittency will experience important changes in community composition.

15 Appendix

15.1 Effects through pathways other than climate change

There are countless effects of climate change on aquatic systems through non-natural pathways – these changes may even be the main cascading effects from climatic drivers that will affect the ecosystems. Lanz et al. (2014) noted that human activities will largely determine the quality of our waters, generally overriding the effects of climate change. Figure 15.1 provides a graphical view of some of the expected impacts, comparing the effects caused directly by climate change and the ones caused by other human activity.

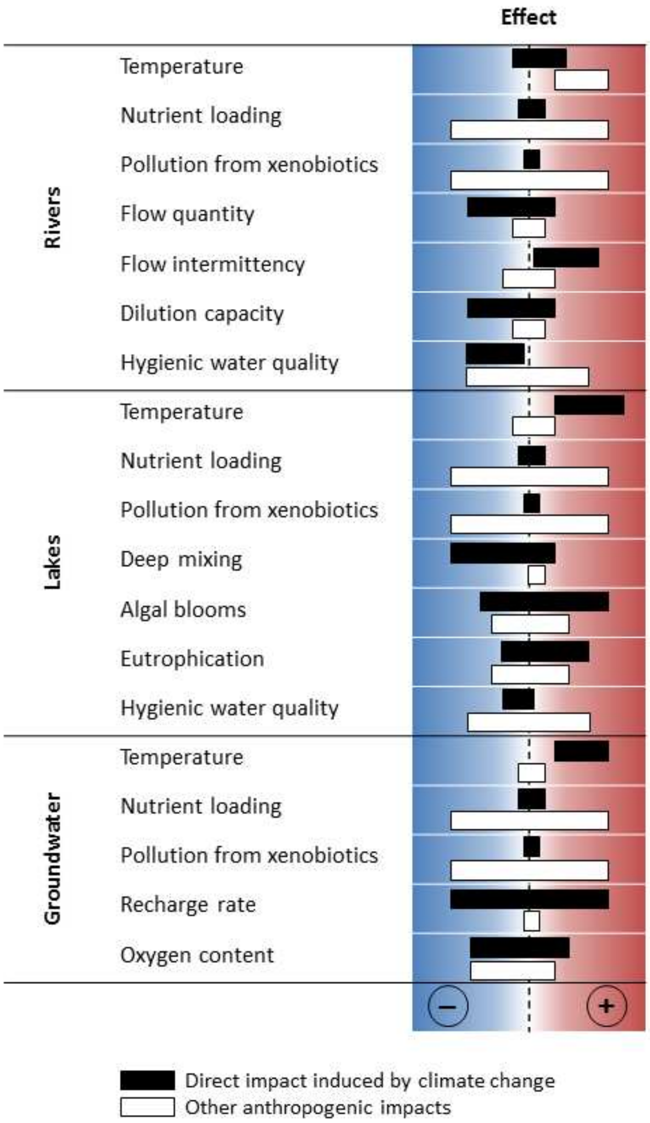


Figure 15.1: Qualitative comparison of the climate-induced impacts and anthropogenic impacts on quantities and processes related to freshwater ecosystems.

These changes are complex and therefore difficult to evaluate. Given their large number and diversity, and the unpredictable character of these changes, we do not review them comprehensively in this work. However, in this section, we will discuss some of these foreseeable changes. They are also often dependent on human activities and therefore on social, economic and political changes.

Land use and agricultural practices will have to adapt to climate change (Akademien der Wissenschaften Schweiz 2016). These changes can strongly affect soil status (Walter and Hänni 2018) and water quality in the affected catchments. The following drivers are relevant:

- **Urbanization** will continue increasing the fraction or urban areas throughout the world, in Switzerland primarily at the expense of agricultural areas, which negatively affects the quality of water resources. These effects can be counteracted by better spatial planning and implementation of adequate measures to reduce chemical exports (M. Schärer, personal communication). Urbanization also impacts water transport: it was shown to significantly increase groundwater recharge rates in a small region in Switzerland (Minnig et al. 2017).

- **Crop changes**, which may be necessary to preserve productivity under future conditions, can affect water and nutrient demand in agriculture (e.g., changes in the distribution of crop types or in the extent of alpine pastures).
- **Restoration of buffers** such as floodplains can help filter nutrients and contaminants before they enter downstream runoff (Lair et al. 2009).
- **Agricultural practices** have the potential to reduce pollution from fields and thereby alleviate some negative impacts of climate change. For example, soil conservation practices can limit increases in erosion and water loss due to climate change (E. Frossard, personal communication; Prasuhn 2012). Similarly, appropriate timing and quantity of irrigation can reduce nutrient leaching while optimizing crop yield (Prasuhn and Albisser 2014). Regarding plant protection products, the Swiss Federal Office for Agriculture launched an action plan in 2017, with the aim of reducing the risks associating with these chemicals.

The transition towards renewable energy sources may increase direct pressure on the environment. Two main drivers should be considered:

- **Increase of hydropower capacity**. Dams affect the aquatic ecosystem by forming barriers and new lakes, and through hydropeaking. The latter affects fish community structure, stranding, reproduction, productivity and macroinvertebrate biomass, diversity and longitudinal zonation (Tonolla et al. 2017). Hydropower use and its interaction with changes in runoff regimes (e.g., due to different seasonal use of electricity due to climate change) are expected to lead to more frequent hydropeaking (i.e., very fast flow rate variations caused by reservoir operation). Hydropeaking strongly influences macrophytes in areas downstream of a dam, because minimum flow is lower than in free-flowing streams. This leads to increased mortality of both floating and submerged aquatic plants during emerged phases (Bejarano et al. 2017). During rapid water level increases, turbulence can remove or damage plant parts, resulting in higher mortality, biomass loss, reduced vigor, and potentially reduced plant fitness (Combroux et al. 2001). These events also affect rooting, through soil scouring and mechanical stress (Lind et al. 2014). Under these conditions, amphibious plants are favored and can dominate communities. These conditions also elicit vegetative reproduction (e.g., runners, rhizomes, vegetative propagules, adventitious root development) (Sorrell et al. 2000). There are, however, important efforts underway mitigate the effects of hydropeaking (see, e.g., Person et al. 2014).
- **Increase of thermal use of waterbodies**. In particular, use of water for cooling purposes is likely to increase in a warmer climate. This would also increase releases of warmer water into waterbodies, especially during summer – a critical season for ecosystems. On the other hand, heat extraction could also increase as an alternative source of heat to fossil fuels – this would instead induce a cooling of the waterbodies, counterbalancing climate-induced effects (Gaudard et al. 2018).

Finally, the technical improvement of wastewater treatment plants may help reduce nutrient and pollutant levels in recipient waterbodies. In addition to improving treatment efficiency, measures should be considered to limit the impact of sewer overflow events, and all the more so if flood frequency increases with climate change.

15.2 Literature search

Searches and extraction of the data were performed with "*Web of Knowledge"* on 04.10.2017 (part Ecological Status) and on 29.03.2018 (part Water Quality). The search was performed as indicated in the following table.

Part	Sections	Search date	Search keywords (TS field)
Water Quality	4–6	29.03.2018	TS=(Freshwater OR Aquatic systems OR lake* OR pond* OR bog* OR stream* OR river* OR freshwater* OR creek* OR lotic OR lentic OR headwater* OR reservoir* OR brook* OR wetland* OR *pool* OR marsh* OR watershed* OR catchment* OR limnol* OR "inland water*") AND TS=("climate change" OR "global change" OR "global warming" OR "climate warming" OR warm* OR heat*) AND TS="water quality"
Ecological Status	7–12	04.10.2017	TS=(Freshwater OR Aquatic systems OR lake* OR pond* OR bog* OR stream* OR river* OR freshwater* OR creek* OR lotic OR lentic OR headwater* OR reservoir* OR brook* OR wetland* OR *pool* OR marsh* OR watershed* OR catchment* OR limnol* OR glacial* OR "inland water*") AND TS=(Switzerland OR Swiss* OR alp* OR mountain*) AND TS=("climate change" OR "global change" OR "global warming" OR warm* OR temperatur* OR heat* OR thermal*)

In order to sort publications by topics, we used the citation network tools "*CitNetExplorer*" and "*VOSviewer*". These tools use the title and abstract information, as well as the cross-citations among all studies, to group studies based on their cross-referencing, creating subgroups of thematically related studies. The final cluster analysis was performed using „*VOSviewer*". We extracted the results and used a text mining analysis on the titles of the publications in order to select the clusters on which systematic review of the titles will be performed. After the filtering based on the titles, a filtering based on the abstracts was performed.

Having selected targeted keywords, we found distinct clusters of studies covering relatively separated study areas. For example, studies looking at effects on (lake) ecology were clearly distinguished from studies which addressed effects on hydrology, studies on rivers, or studies on past (paleological) climate change. The latter, for example, were mostly not relevant for our work. Using these clusters (see Figure 15.2), we could identify further papers, or, by looking at cross-references, ensure that we did not have major thematic gaps.

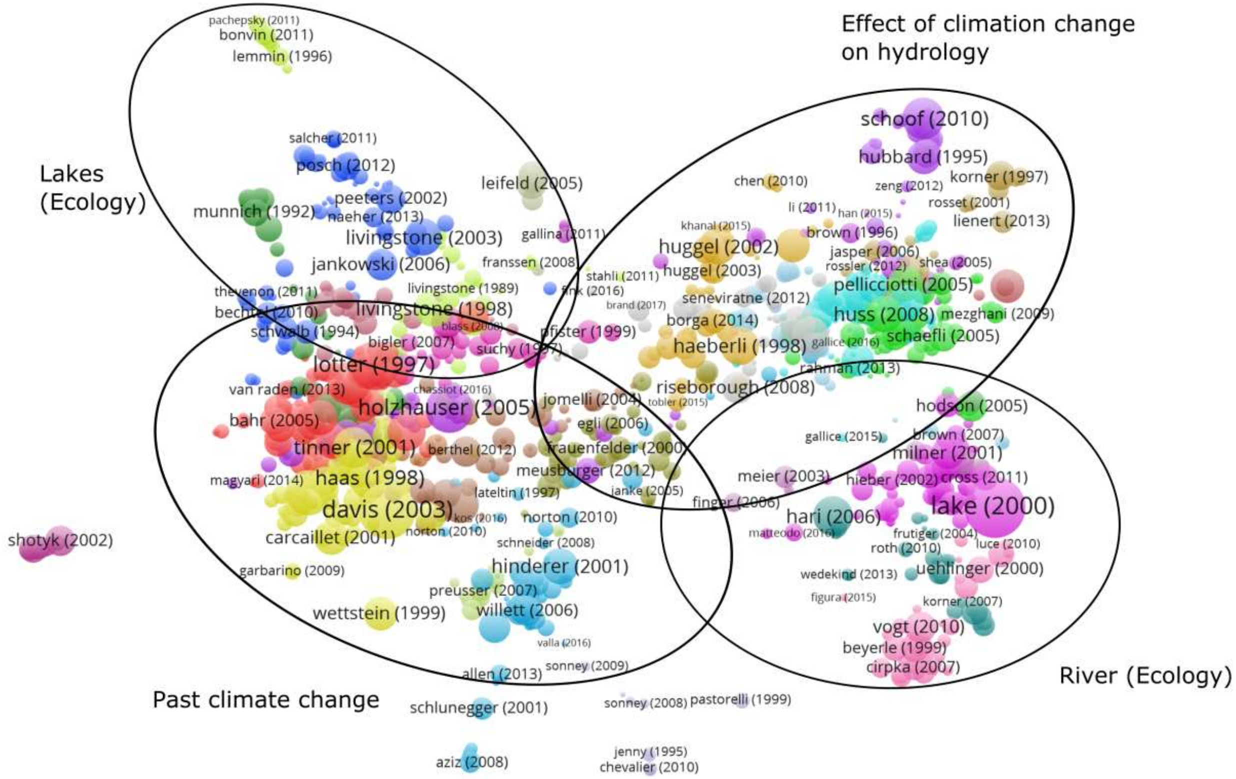


Figure 15.2: Example map of the citation network and annotations for a search (part: Ecological Status) with focus on Switzerland. Colors represent clusters; the size of the points is proportional to the number citations of the articles. The main regions of the plot are identified by ellipses and labeled.

In order to identify the level of coverage for each part, we extracted from all the documents/studies cited in the report the method, the area, and time period (year) of study. We visualized the ratio of studies from Switzerland relative to studies from other areas, in order to have an estimation of the coverage of Swiss-specific studies for each section (see Figure 15.3).

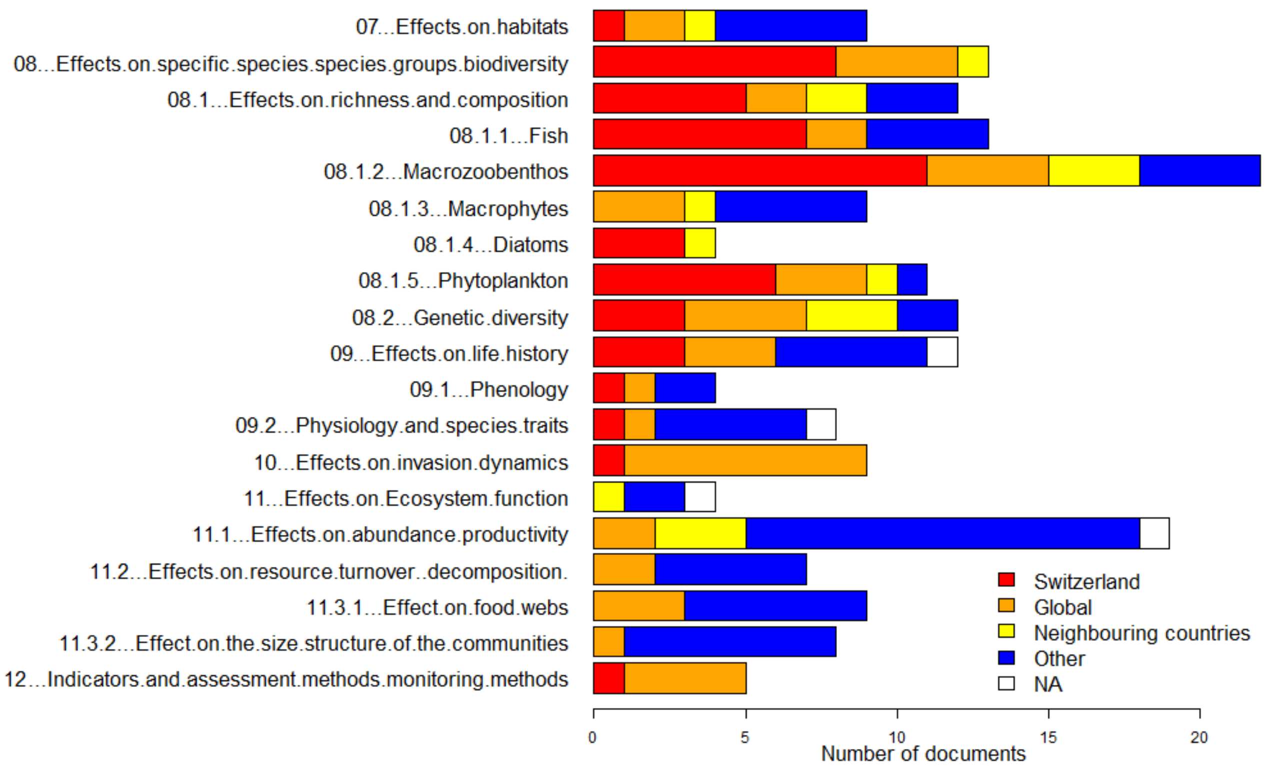


Figure 15.3: Staggered barplot of the references used in the different sections (part: Ecological Status). Colors indicates the area where the studies were conducted: Switzerland (red), global (orange), neighboring countries (yellow), and other countries (blue).

The following table lists the literature cited in Sections 5 and 6 (part: Water Quality), providing a unified view of the characteristics of these references.

Chapter	Reference	Methods	Theme	Sections	Focus area	Focus catchment type
5.1	(Edwards et al. 2007)	Review	Impact of snow cover on soils	5.1, 5.2, 5.6		Alpine
	(Evans et al. 2005)	Observations	DOC and acidity in surface waters	5.1, 5.2, 6.1, 6.2, 6.3	UK	

	(Rodríguez-Murillo et al. 2015)	Observations	OC in rivers	5.1	Switzerland	
	(Rodríguez-Murillo and Filella 2015)	Observations	OC in lakes	5.1	Switzerland	
	(Hagedorn et al. 2018)	Review		5.1, 5.5	Switzerland	
	(Schimel et al. 2007)	General article	Microbial community	5.1		
	(Davidson and Janssens 2006)	Review	Soil carbon decomposition	5.1		
	(Kirschbaum 1995)	Review	OC in soils	5.1		
	(Johnston et al. 2004)	Review	Carbon cycling in soils	5.1		
	(Kalbitz et al. 2000)	Review	DOM in soils	5.1, 5.2, 5.5, 5.6		
	(Bardgett et al. 2008)	General article	Microbes and carbon cycle	5.1, 5.2		
	(Schlesinger et al. 2015)	General article	Forest response to drought	5.1, 5.2, 5.5		Forest
	(Boxall et al. 2009)	Review	Chemicals cycling	5.1, 5.2, 5.4, 6.1		Agricultural
	(Matzner and Borken 2008)	Review	Nutrient cycling in soils	5.1, 5.2		
	(Bernal et al. 2012)	Observations	Nitrogen cycle	5.1, 5.2, 5.5, 5.6		Forest
	(Groffman et al. 2001)	Observations	Carbon and nutrient cycling in freeze-thaw events	5.1	USA	Forest
	(Callesen et al. 2007)	Observations	Nitrogen leaching in freeze-thaw events	5.1, 5.2	Europe	Forest
	(Noyes et al. 2009)	Review	Contaminants	5.1, 5.2, 5.4, 5.6		
	(Kallenborn et al. 2012)	Review	POPs	5.1, 5.2, 5.6		
5.2	(Scheurer et al. 2009)	Review	Climate change, erosion and river sediment	5.2	Alpine countries	
	(Steingruber and Colombo 2006)	Observations	Surface water acidification	5.2	Switzerland	Alpine
	(Prasuhn and Albisser 2014)	Observations	Irrigation and nitrogen leaching	5.2	Switzerland	Agricultural

	(Carpenter et al. 2017)	Observations	Phosphorus cycle	5.2	USA	Agricultural
	(Skjelkvåle et al. 2003)	General article	Acidification of surface waters	5.2	Europe	
	(Fitzhugh et al. 2001)	Observations	Carbon and nutrient cycling in freeze-thaw events	5.2	USA	Forest
	(Bloomfield et al. 2006)	Review	Pesticides	5.2, 5.4	UK	
	(Gurung and Stähli 2014)	Review		5.1, 5.2, 5.4, 6.1, 6.3	Switzerland	
	(Hoffmann et al. 2014)	Review		5.2, 6.1, 6.2, 6.3	Switzerland	
	(Lin et al. 2001)	Modelling	Nitrate leaching	5.2, 5.4, 5.6		Agricultural
	(Murdoch et al. 2000)	Review	Water quality	5.2, 5.4, 6.2	North America	
	(Jeppesen et al. 2009)	General article	Lake dynamics, phosphorus cycle, runoff	5.2, 6.2	Europe	
	(Royer et al. 2006)	Observations	Nutrient loading in rivers	5.2, 5.4	USA	Agricultural
	(Sharpley et al. 2001)	General article	Phosphorus transport	5.2, 5.4	USA	Agricultural
	(Schindler 1997)	Review	Water quality	5.2, 6.1, 6.2	North America	
	(Rossi and Hari 2009)	Observations	Urban stormwater and temperature	5.3	Switzerland	Urban
5.3	(Rossi et al. 2004)	Observations	Urban stormwater and PCBs	5.3	Switzerland	Urban
	(Rossi et al. 2013)	Observations	Sediment pollution in urban streams	5.3	Switzerland	Urban
	(Coutu et al. 2013)	Modelling	Urban runoff pollution and climate change	5.3	Switzerland	Urban
	(Buerge et al. 2006)	Observations	Combined sewer overflow in lake	5.3	Switzerland	Agricultural and urban
	(Herb et al. 2008)	Modelling	Thermal pollution	5.3	USA	Urban
	(Nie et al. 2009)	Modelling	Urban drainage systems	5.3	Norway	Urban
	(Fortier and Mailhot 2015)	Modelling	Combined sewer overflows	5.3	Canada	Urban
	(Patz et al. 2008)	Modelling	Combined sewer overflows	5.3	USA	
	(Milly et al. 2002)	Observations and modelling	Extreme floods	5.3		

	(Heinz et al. 2009)	Observations	Sewage pollution	5.3, 6.1	Germany	Urban
5.4	(Huntscha et al. 2018)	Observations and modelling	Pesticides lake	5.4	Switzerland	Agricultural
	(IPCC 2007)	Review		5.4		
	(Donald et al. 2007)	Observations	Pesticides	5.4	North America	
	(Flury 1996)	Review	Pesticides transport	5.4		
	(Foley et al. 2012)	Case study	Lake dynamics	5.4	UK	
	(Besmer et al. 2017)	Observations	Karst spring water quality	5.4	Switzerland	
	(Fuhrer et al. 2006)	General article	Climate change in agriculture and forests	5.4, 5.5	Switzerland	Agricultural and forest
	(Arheimer et al. 2005)	Modelling	Water quality	5.1, 5.4, 6.1, 6.2	Sweden	
	(Lindner et al. 2010)	Review	Forest ecosystems	5.5, 5.6	Europe	Forest
5.5	(Schindler 2009)	General article	Lakes dynamics	5.5, 6.1, 6.2		
	(Campbell et al. 2000)	Observations	Nitrogen cycle	5.6	USA	Alpine
5.6	(Dawes et al. 2016)	Observations (experimental warming)	Nitrogen cycling in soils	5.6	Switzerland	Alpine
	(Streit et al. 2013)	Observations (experimental warming)	Carbon cycling in soils	5.6	Switzerland	Alpine
	(Köplin et al. 2014)	Modelling	Floods dynamics	5.6	Switzerland	
	(Bavay et al. 2013)	Modelling	Climate change and snow cover	5.6	Switzerland	Alpine
	(Hiltbrunner et al. 2005)	Observations	N storage in snow	5.6	Switzerland	Alpine
	(Morán-Tejeda et al. 2016)	Observations	Rain-on-snow events	5.6	Switzerland	Alpine
	(Hruška et al. 2009)	Observations	DOC in streams	5.6	Central Europe	Forest
	(Jenkins et al. 1993)	Observations	Snowmelt	5.6	Scotland	Alpine
	(Sadro et al. 2018)	Observations	Snowmelt and lake dynamics	5.6, 6.1	USA	Alpine
	(Costa et al. 2018)	Modelling	Sediment export	5.6, 6.1	Switzerland	Alpine

	(Wang et al. 1999)		Water resources	5.6, 6.1	Alaska	
	(Rouse et al. 1997)	Review	Water quality	5.6, 6.1, 6.2	North America	Arctic
	(Monteith et al. 2007)	Observations	DOC deposition	5.6, 6.3		
	(Hejzlar et al. 2003)	Observations and modelling	DOM in streams	6.1	Czech Republic	
6.1	(Boano et al. 2014)	Review	Hyporheic transport processes	6.1		
	(Ulseth et al. 2018)	Observations	Snowmelt and stream carbon dynamics	6.1	Austria	Alpine
	(Ducharne 2008)	Modelling	Stream dynamics	6.1	France	
	(Boissier et al. 1996)	Observations	Drought and river-groundwater exchange	6.1	France	
	(Cruise et al. 1999)	Observations and modelling	Water quality	6.1	USA	
	(Woolway et al. 2018)	Observations	Lake dynamics	6.1	England	
	(Mulholland et al. 1997)	General article	Freshwater ecosystems	6.1, 6.2	USA, Mexico	
	(Magnuson et al. 1997)	Review	Lakes dynamics	6.1, 6.2	Canada	
	(Bader et al. 2004)	Observations	Effect of dry and hot summer (2003)	6.1, 6.3	Switzerland	
	(Zobrist et al. 2018)	Observations	River chemistry	6.1, 6.3	Switzerland	
6.2	(Porter et al. 1996)	Case study	Lake dynamics	6.2	Italy	
	(Råman Vinnå et al. 2018)	Modelling	Lake dynamics	6.2	Switzerland	
	(Koinig et al. 1998)	Case study	Lake dynamics	6.2	Austria	Alpine
	(Sommaruga-Wögrath et al. 1997)	Observations	Lake dynamics	6.2	Austria	Alpine
	(Yankova et al. 2017)	Observations	Lake dynamics	6.2	Switzerland	
	(Isles et al. 2017)	Observations	Lake dynamics	6.2	USA, Canada	
	(Weyhenmeyer et al. 2007)	Observations	Lake dynamics	6.2	Europe	
	(Schwefel et al. 2016)	Observations and modelling	Lake dynamics	6.2	Switzerland	

	(Gudasz et al. 2010)	Observations		6.2	Sweden	
	(Perga et al. 2018)	Observations and modelling	Lake dynamics	6.2	France	Alpine
	(Moore et al. 1997)	General article	Freshwater ecosystems	6.2	Mid-Atlantic	
	(Paerl 2014)	General article	Cyanobacteria	6.2		
	(Wood et al. 2017)	Observations	Cyanobacteria	6.2	New Zealand	
	(Paerl and Huisman 2008)	General article	Algal blooms	6.2		
	(Salmaso et al. 2017)	Case study	Cyanobacteria	6.2	Italy	
	(Walter and Hänni 2018)	(nfp68)		6.3	Switzerland	
6.3	(Figura et al. 2013)	Observations	River-fed groundwater dynamics	6.3	Switzerland	
	(IPCC 2014)	(ipcc)				
	(FOEN 2009)	Observations	Groundwater monitoring	6.3	Switzerland	
	(FOEN 2016b)	Observations	Effect of dry and hot summer (2015)	6.3	Switzerland	
	(Jeannin et al. 2016)	Observations	Groundwater mineralization	6.3	Switzerland	
	(FOEN 2019)	Observations	Groundwater monitoring	6.3	Switzerland	
Other	(Bates et al. 2008)	Review				
	(Minnig et al. 2017)	Observations	Groundwater recharge	* Urbanization increased groundwater recharge rates	Switzerland	Urban
	(Akademien der Wissen-schaften Schweiz 2016)				Switzerland	
	(Bouraoui et al. 2004)	Modelling	Hydrological and nutrient cycle		Finland	
	(Lair et al. 2009)	Review	Fate of contaminants in rivers		Europe	
	(Fink et al. 2014)	Modelling	Lake dynamics		Switzerland	
	(Honti et al. 2017)	Modelling	Catchment management			

15.3 Expert interviews

The following table lists the experts interviewed in the framework of this work.

Date	Name(s)	Affiliation	Department
03.11.2017	Francesco Pomati	Eawag	Aquatic Ecology
03.11.2017	Nele Schuwirth	Eawag	Systems Analysis, Integrated Assessment and Modelling
06.11.2017	Jakob Brodersen	Eawag	Fish Ecology and Evolution
06.11.2017	Alfred Wüest	Eawag	Surface Waters Research and Management
07.11.2017	Christopher Robinson	Eawag	Aquatic Ecology
08.11.2017	Otto Seppälä	Eawag	Aquatic Ecology
09.11.2017	Christine Weber	Eawag	Surface Waters Research and Management
18.04.2018	Mario Schirmer	Eawag	Water Resources and Drinking Water
03.05.2018	Marianne Balmer, Thomas Poiger	Agroscope	Plants and Plant Products
08.05.2018	Klaus Lanz	International Water Affairs	
09.05.2018	Urs von Gunten	Eawag	Water Resources & Drinking Water
15.05.2018	Volker Prasuhn, Walter Richner	Agroscope	Agroecology and Environment
18.05.2018	Jürg Zobrist	Eawag	Surface Waters Research and Management
28.05.2018	Pius Niederhauser	Canton Zürich	Water Protection
28.05.2018	Emmanuel Frossard	ETHZ	Agricultural Sciences
31.05.2018	Frank Hagedorn	WSL	Forest Soils and Biogeochemistry
11.06.2018	Nathalie Chèvre	UniL	Earth Surface Dynamics
14.06.2018	Michael Schärer	FOEN	Water Protection
07.09.2018	Max Maurer	Eawag	Urban Water Management
05.12.2018	Adriano Joss	Eawag	Wastewater Management

16 References

Abegg M., Brändli U.-B., Cioldi F., Fischer C., Herold-Bonardi A., Huber M., Keller M., Meile R., Rösler E., Speich S., Traub B., Vidondo B. 2014: Fourth national forest inventory - result tables and maps on the Internet for the NFI 2009-2013 (NFI4b). Swiss Federal Research Institute WSL, Birmensdorf.

Adams G.L., Pichler D.E., Cox E.J., O'Gorman E.J., Seeney A., Woodward G., Reuman D.C. 2013: Diatoms can be an important exception to temperature–size rules at species and community levels of organization. Glob. Change Biol. Vol. 19, No. 11: 3540–3552.

Akademien der Wissenschaften Schweiz 2016: Brennpunkt Klima Schweiz: Grundlagen, Folgen und Perspektiven. Swiss Academies Reports. Akademien der Wissenschaften Schweiz, ProClim, Bern.

Altermatt F. 2010a: Climatic warming increases voltinism in European butterflies and moths. Proc. R. Soc. B Biol. Sci. Vol. 277, No. 1685: 1281–1287.

Altermatt F. 2010b: Tell me what you eat and I'll tell you when you fly: diet can predict phenological changes in response to climate change. Ecol. Lett. Vol. 13: 1475–1484.

Altermatt F., Seymour M., Martinez N. 2013: River network properties shape α-diversity and community similarity patterns of aquatic insect communities across major drainage basins. J. Biogeogr. Vol. 40, No. 12: 2249–2260.

Alther R., Altermatt F. 2018: Fluvial network topology shapes communities of native and non-native amphipods. Ecosphere Vol. 9, No. 2: e02102.

Amalfitano S., Coci M., Corno G., Luna G.M. 2015: A microbial perspective on biological invasions in aquatic ecosystems. Hydrobiologia Vol. 746, No. 1: 13–22.

Ambrosetti W., Barbanti L. 1999: Deep water warming in lakes: An indicator of climatic change. J. Limnol. Vol. 58, No. 1: 1–9.

Andersen H.E., Kronvang B., Larsen S.E., Hoffmann C.C., Jensen T.S., Rasmussen E.K. 2006: Climate-change impacts on hydrology and nutrients in a Danish lowland river basin. Sci. Total Environ., Monitoring and modelling the impacts of global change on European freshwater ecosystems Vol. 365, No. 1: 223–237.

Angermeier P.L. 1995: Ecological Attributes of Extinction-Prone Species: Loss of Freshwater Fishes of Virginia. Conserv. Biol. Vol. 9, No. 1: 143–158.

Arheimer B., Andréasson J., Fogelberg S., Johnsson H., Pers C.B., Persson K. 2005: Climate Change Impact on Water Quality: Model Results from Southern Sweden. AMBIO J. Hum. Environ. Vol. 34, No. 7: 559–566.

Atwood T.B., Hammill E., Kratina P., Greig H.S., Shurin J.B., Richardson J.S. 2015: Warming alters food web-driven changes in the CO2 flux of experimental pond ecosystems. Biol. Lett. Vol. 11, No. 12: 20150785.

Bader S., Devanthéry D., Hertig A., Jakob A., Kan C., Kohli E., Kozel R., Kummer M., Liechti H., Liechti P., Lukes R., Marti P., Roulier C., Schorer M., Schürch M., Sieber U. 2004: Auswirkungen des Hitzesommers 2003 auf die Gewässer. Schriftenreihe Umwelt. BUWAL, BWG, MeteoSchweiz, Bern.

Bálint M., Domisch S., Engelhardt C.H.M., Haase P., Lehrian S., Sauer J., Theissinger K., Pauls S.U., Nowak C. 2011: Cryptic biodiversity loss linked to global climate change. Nat. Clim. Change Vol. 1, No. 6: 313–318.

Bardgett R.D., Freeman C., Ostle N.J. 2008: Microbial contributions to climate change through carbon cycle feedbacks. ISME J. Vol. 2, No. 8: 805–814.

Barnes K.B., Morgan J.M., Roberge M.C. 2002: Impervious surfaces and the quality of natural and built environments. Towson University, Baltimore.

Baroiller J.F., D'Cotta H., Saillant E. 2009: Environmental Effects on Fish Sex Determination and Differentiation. Sex. Dev. Vol. 3, No. 2–3: 118–135.

Bates B., Kundzewicz Z., Wu S. 2008: Climate Change and Water. Intergovernmental Panel on Climate Change Secretariat, Geneva, Switzerland.

Bavay M., Grünewald T., Lehning M. 2013: Response of snow cover and runoff to climate change in high Alpine catchments of Eastern Switzerland. Adv. Water Resour., Snow–Atmosphere Interactions and Hydrological Consequences Vol. 55: 4–16.

BDM Coordination Office 2014: Swiss Biodiversity Monitoring BDM. Description of Methods and Indicators, Bern.

Bejarano M.D., Jansson R., Nilsson C. 2017: The effects of hydropeaking on riverine plants: a review. Biol. Rev. Vol. 93, No. 1: 658–673.

Bellard C., Thuiller Wilfried, Leroy Boris, Genovesi Piero, Bakkenes Michel, Courchamp Franck 2013: Will climate change promote future invasions? Glob. Change Biol. Vol. 19, No. 12: 3740–3748.

Bellmore J.R., Duda J.J., Craig L.S., Greene S.L., Torgersen C.E., Collins M.J., Vittum K. 2017: Status and trends of dam removal research in the United States. Wiley Interdiscip. Rev. Water Vol. 4, No. 2: e1164.

Bernal S., Hedin L.O., Likens G.E., Gerber S., Buso D.C. 2012: Complex response of the forest nitrogen cycle to climate change. Proc. Natl. Acad. Sci. Vol. 109, No. 9: 3406–3411.

Besmer M.D., Hammes F., Sigrist J.A., Ort C. 2017: Evaluating Monitoring Strategies to Detect Precipitation-Induced Microbial Contamination Events in Karstic Springs Used for Drinking Water. Front. Microbiol. Vol. 8: 2229.

Beutel M.W., Horne A.J. 2018: Nutrient Fluxes From Profundal Sediment of Ultra-oligotrophic Lake Tahoe, California/Nevada: Implications for Water Quality and Management in a Changing Climate. Water Resour. Res. Vol. 54, No. 3: 1549–1559.

Bloomfield J.P., Williams R.J., Gooddy D.C., Cape J.N., Guha P. 2006: Impacts of climate change on the fate and behaviour of pesticides in surface and groundwater—a UK perspective. Sci. Total Environ. Vol. 369, No. 1: 163–177.

Boano F., Harvey J.W., Marion A., Packman A.I., Revelli R., Ridolfi L., Wörman A. 2014: Hyporheic flow and transport processes: Mechanisms, models, and biogeochemical implications. Rev. Geophys. Vol. 52, No. 4: 2012RG000417.

Boissier J.-M., Marmonier P., Claret C., Fontvieille D., Blanc P. 1996: Comparison of solutes, nutrients, and bacteria inputs from two types of groundwater to the Rhône river during an artificial drought. Hydrobiologia Vol. 319, No. 1: 65–72.

Borsuk M.E., Reichert P., Peter A., Schager E., Burkhardt-Holm P. 2006: Assessing the decline of brown trout (Salmo trutta) in Swiss rivers using a Bayesian probability network. Ecol. Model. Vol. 192, No. 1: 224–244.

Bouraoui F., Grizzetti B., Granlund K., Rekolainen S., Bidoglio G. 2004: Impact of Climate Change on the Water Cycle and Nutrient Losses in a Finnish Catchment. Clim. Change Vol. 66, No. 1–2: 109–126.

Boxall A.B.A., Hardy A., Beulke S., Boucard T., Burgin L., Falloon P.D., Haygarth P.M., Hutchinson T., Kovats R.S., Leonardi G., Levy L.S., Nichols G., Parsons S.A., Potts L., Stone D., Topp E., Turley D.B., Walsh K., Wellington E.M.H., Williams R.J. 2009: Impacts of Climate Change on Indirect Human Exposure to Pathogens and Chemicals from Agriculture. Environ. Health Perspect. Vol. 117, No. 4: 508–514.

Boyero L., Pearson R.G., Gessner M.O., Barmuta L.A., Ferreira V., Graça M.A.S., Dudgeon D., Boulton A.J., Callisto M., Chauvet E., Helson J.E., Bruder A., Albariño R.J., Yule C.M., Arunachalam M., Davies J.N., Figueroa R., Flecker A.S., Ramírez A., Death R.G., Iwata T., Mathooko J.M., Mathuriau C., Gonçalves J.F., Moretti M.S., Jinggut T., Lamothe S., M'Erimba C., Ratnarajah L., Schindler M.H., Castela J., Buria L.M., Cornejo A., Villanueva V.D., West D.C. 2011: A global experiment suggests climate warming will not accelerate litter decomposition in streams but might reduce carbon sequestration. Ecol. Lett. Vol. 14, No. 3: 289–294.

Brodersen J., Seehausen O. 2014: Why evolutionary biologists should get seriously involved in ecological monitoring and applied biodiversity assessment programs. Evol. Appl. Vol. 7, No. 9: 968–983.

Brooks R.T. 2004: Weather-related effects on woodland vernal pool hydrology and hydroperiod. Wetlands Vol. 24, No. 1: 104–114.

Brown J.H., Gillooly J.F., Allen A.P., Savage V.M., West G.B. 2004: Toward a metabolic theory of ecology. Ecology Vol. 85, No. 7: 1771–1789.
Brown L.E., Dickson N.E., Carrivick J.L., Füreder L. 2015: Alpine river ecosystem response to glacial and anthropogenic flow pulses. Freshw. Sci. Vol. 34, No. 4: 1201–1215.
Brown L.E., Hannah D.M., Milner A.M. 2007: Vulnerability of alpine stream biodiversity to shrinking glaciers and snowpacks. Glob. Change Biol. Vol. 13, No. 5: 958–966.
Brown L.E., Hannah D.M., Milner A.M. 2003: Alpine Stream Habitat Classification: An Alternative Approach Incorporating the Role of Dynamic Water Source Contributions. Arct. Antarct. Alp. Res. Vol. 35, No. 3: 313–322.
Brown L.E., Milner A.M. 2012: Rapid loss of glacial ice reveals stream community assembly processes. Glob. Change Biol. Vol. 18, No. 7: 2195–2204.
Buerge I.J., Poiger T., Müller M.D., Buser H.-R. 2006: Combined Sewer Overflows to Surface Waters Detected by the Anthropogenic Marker Caffeine. Environ. Sci. Technol. Vol. 40, No. 13: 4096–4102.
Burgherr P., Ward J.V. 2001: Longitudinal and seasonal distribution patterns of the benthic fauna of an alpine glacial stream (Val Roseg, Swiss Alps). Freshw. Biol. Vol. 46, No. 12: 1705–1721.
Burkhardt-Holm P., Peter A., Segner H. 2002: Decline of fish catch in Switzerland. Aquat. Sci. Vol. 64, No. 1: 36–54.
Caesar L., Rahmstorf S., Robinson A., Feulner G., Saba V. 2018: Observed fingerprint of a weakening Atlantic Ocean overturning circulation. Nature Vol. 556, No. 7700: 191–196.
Caissie D. 2006: The thermal regime of rivers: A review. Freshw. Biol. Vol. 51: 1389–1406.
Callesen I., Borken W., Kalbitz K., Matzner E. 2007: Long-term development of nitrogen fluxes in a coniferous ecosystem: Does soil freezing trigger nitrate leaching? J. Plant Nutr. Soil Sci. Vol. 170, No. 2: 189–196.
Campbell D.H., Baron J.S., Tonnessen K.A., Brooks P.D., Schuster P.F. 2000: Controls on nitrogen flux in alpine/subalpine watersheds of Colorado. Water Resour. Res. Vol. 36, No. 1: 37–47.
Canning E.U., Curry A., Feist S.W., Longshaw M., Okamura B. 2000: A new class and order of myxozoans to accommodate parasites of bryozoans with ultrastructural observations on Tetracapsula bryosalmonae. J. Eukaryot. Microbiol. Vol. 47: 456–468.
Cardinale B.J., Duffy J.E., Gonzalez A., Hooper D.U., Perrings C., Venail P., Narwani A., Mace G.M., Tilman D., Wardle D.A., Kinzig A.P., Daily G.C., Loreau M., Grace J.B., Larigauderie A., Srivastava D.S., Naeem S. 2012: Biodiversity loss and its impact on humanity. Nature Vol. 486, No. 7401: 59–67.
Carpenter S.R., Booth E.G., Kucharik C.J. 2017: Extreme precipitation and phosphorus loads from two agricultural watersheds. Limnol. Oceanogr. Vol. 63, No. 3: 1221–1233.
Carraro L., Bertuzzo E., Mari L., Fontes I., Hartikainen H., Strepparava N., Schmidt-Posthaus H., Wahli T., Jokela J., Gatto M., Rinaldo A. 2017: Integrated field, laboratory, and theoretical study of PKD spread in a Swiss prealpine river. Proc. Natl. Acad. Sci. Vol. 114, No. 45: 11992–11997.
Cauvy-Fraunié S., Andino P., Espinosa R., Calvez R., Jacobsen D., Dangles O. 2016: Ecological responses to experimental glacier-runoff reduction in alpine rivers. Nat. Commun. Vol. 7: 12025.
CH2011 2011: Swiss climate change scenarios CH2011. C2SM, MeteoSwiss, ETH, NCCR Climate, and OcCC, Zürich Switzerland.
CH2014-Impacts 2014: Toward Quantitative Scenarios of Climate Change Impacts in Switzerland. OcCR, FOEN, MeteoSwiss, C2SM, Agroscope and ProClim, Bern, Switzerland.
Cianfrani C., Satizábal H.F., Randin C. 2015: A spatial modelling framework for assessing climate change impacts on freshwater ecosystems: Response of brown trout (Salmo trutta L.) biomass to warming water temperature. Ecol. Model. Vol. 313, No. Supplement C: 1–12.

CIPAIS 2016a: Pannello di controllo sullo stato e sull'evoluzione delle acque del Lago di Lugano - Anno 2016. Commissione Internazionale per la Protezione delle Acque Italo - Svizzere.
CIPAIS 2016b: Pannello di controllo sullo stato e sull'evoluzione delle acque del Lago Maggiore - Anno 2016. Commissione Internazionale per la Protezione delle Acque Italo - Svizzere.
Cohen J.M., Lajeunesse M.J., Rohr J.R. 2018: A global synthesis of animal phenological responses to climate change. Nat. Clim. Change: 1.
Collins S.M., Kohler Tyler J., Thomas Steven A., Fetzer William W., Flecker Alexander S. 2015: The importance of terrestrial subsidies in stream food webs varies along a stream size gradient. Oikos Vol. 125, No. 5: 674–685.
Combroux I., Bornette G., Willby N., Amoros C. 2001: Regenerative strategies of aquatic macrophytes in flood disturbed habitats : the role of the propagule bank. Arch. Hydrobiol. Vol. 152: 215–235.
Comte L., Grenouillet G. 2013: Do stream fish track climate change? Assessing distribution shifts in recent decades. Ecography Vol. 36, No. 11: 1236–1246.
Conti L., Schmidt-Kloiber A., Grenouillet G., Graf W. 2014: A trait-based approach to assess the vulnerability of European aquatic insects to climate change. Hydrobiologia Vol. 721, No. 1: 297–315.
Costa A., Molnar P., Stutenbecker L., Bakker M., Silva T.A., Schlunegger F., Lane S.N., Loizeau J.-L., Girardclos S. 2018: Temperature signal in suspended sediment export from an Alpine catchment. Hydrol Earth Syst Sci Vol. 22, No. 1: 509–528.
Coutu S., Kramer S., Bates B., Roudier P. 2013: Assessing dominant uncertainties in urban buildup/washoff processes under climate change: A case study in Western Switzerland. Urban Clim. Vol. 5: 52–67.
Craig J.K., Foote C.J., Wood C.C. 1996: Evidence for temperature-dependent sex determination in sockeye salmon (Oncorhynchus nerka). Can. J. Fish. Aquat. Sci. Vol. 53, No. 1: 141–147.
Crozier L.G., Hendry A.P., Lawson P.W., Quinn T.P., Mantua N.J., Battin J., Shaw R.G., Huey R.B. 2008: Potential responses to climate change in organisms with complex life histories: evolution and plasticity in Pacific salmon. Evol. Appl. Vol. 1, No. 2: 252–270.
Cruise J.F., Limaye A.S., Al-Abed N. 1999: Assessment of Impacts of Climate Change on Water Quality in the Southeastern United States. JAWRA J. Am. Water Resour. Assoc. Vol. 35, No. 6: 1539–1550.
Davidson E.A., Janssens I.A. 2006: Temperature sensitivity of soil carbon decomposition and feedbacks to climate change. Nature Vol. 440, No. 7081: 165.
Dawes M.A., Schleppi P., Hättenschwiler S., Rixen C., Hagedorn F. 2016: Soil warming opens the nitrogen cycle at the alpine treeline. Glob. Change Biol. Vol. 23, No. 1: 421–434.
Demars B.O. I., Russell Manson J., Ólafsson J.S., Gíslason G.M., Gudmundsdóttir R., Woodward G., Reiss J., Pichler D.E., Rasmussen J.J., Friberg N. 2011: Temperature and the metabolic balance of streams. Freshw. Biol. Vol. 56, No. 6: 1106–1121.
Dingemanse N.J., Kalkman V.J. 2008: Changing temperature regimes have advanced the phenology of Odonata in the Netherlands. Ecol. Entomol. Vol. 33, No. 3: 394–402.
Dodds W.K., Perkin J.S., Gerken J.E. 2013: Human Impact on Freshwater Ecosystem Services: A Global Perspective. Environ. Sci. Technol. Vol. 47, No. 16: 9061–9068.
Dokulil M.T., Jagsch A., George G.D., Anneville O., Jankowski T., Wahl B., Lenhart B., Blenckner T., Teubner K. 2006: Twenty years of spatially coherent deepwater warming in lakes across Europe related to the North Atlantic Oscillation. Limnol. Oceanogr. Vol. 51, No. 6: 2787–2793.
Donald D.B., Cessna A.J., Sverko E., Glozier N.E. 2007: Pesticides in Surface Drinking-Water Supplies of the Northern Great Plains. Environ. Health Perspect. Vol. 115: 1183–1191.
Doppler T., Mangold S., Wittmer I., Spycher S., Comte R., Stamm C., Singer H., Junghans M., Kunz M. 2017: Hohe PSM-Belastung in Schweizer Bächen - NAWA-SPEZ-Kampagne untersucht Bäche in Gebieten intensiver landwirtschaftlicher Nutzung. Aqua Gas No. 4: 46–56.

Dossena M., Yvon-Durocher G., Grey J., Montoya J.M., Perkins D.M., Trimmer M., Woodward G. 2012: Warming alters community size structure and ecosystem functioning. Proc. R. Soc. Lond. B Biol. Sci. Vol. 279, No. 1740: 3011–3019.
Dragoni W., Sukhija B.S. 2008: Climate change and groundwater: a short review. Geol. Soc. Lond. Spec. Publ. Vol. 288, No. 1: 1–12.
Driscoll D.A., Felton A., Gibbons P., Felton A.M., Munro N.T., Lindenmayer D.B. 2012: Priorities in policy and management when existing biodiversity stressors interact with climate-change. Clim. Change Vol. 111, No. 3–4: 533–557.
Ducharne A. 2008: Importance of stream temperature to climate change impact on water quality. Hydrol Earth Syst Sci Vol. 12, No. 3: 797–810.
Dudgeon D., Arthington A.H., Gessner M.O., Kawabata Z.-I., Knowler D.J., Lévêque C., Naiman R.J., Prieur-Richard A.-H., Soto D., Stiassny M.L.J., Sullivan C.A. 2006: Freshwater biodiversity: importance, threats, status and conservation challenges. Biol. Rev. Vol. 81, No. 2: 163–182.
Eby L.A., Helmy O., Holsinger L.M., Young M.K. 2014: Evidence of Climate-Induced Range Contractions in Bull Trout Salvelinus confluentus in a Rocky Mountain Watershed, U.S.A. PLOS ONE Vol. 9, No. 6: e98812.
Eckhardt K., Ulbrich U. 2003: Potential impacts of climate change on groundwater recharge and streamflow in a central European low mountain range. J. Hydrol. Vol. 284, No. 1: 244–252.
Edwards A.C., Scalenghe R., Freppaz M. 2007: Changes in the seasonal snow cover of alpine regions and its effect on soil processes: A review. Quat. Int., The Soil Record of Quaternary Climate Change Vol. 162–163: 172–181.
Engering A., Hogerwerf L., Slingenbergh J. 2013: Pathogen–host–environment interplay and disease emergence. Emerg. Microbes Infect. Vol. 2, No. 2: e5.
Etter S., Addor N., Huss M., Finger D. 2017: Climate change impacts on future snow, ice and rain runoff in a Swiss mountain catchment using multi-dataset calibration. J. Hydrol. Reg. Stud. Vol. 13, No. Supplement C: 222–239.
Evans C.D., Monteith D.T., Cooper D.M. 2005: Long-term increases in surface water dissolved organic carbon: Observations, possible causes and environmental impacts. Environ. Pollut., Recovery from acidification in the UK: Evidence from 15 years of acid waters monitoring Vol. 137, No. 1: 55–71.
Everall N.C., Johnson M.F., Wilby R.L., Bennett C.J. 2015: Detecting phenology change in the mayfly *Ephemera danica*: responses to spatial and temporal water temperature variations: Mayfly phenology in relation to river temperature. Ecol. Entomol. Vol. 40, No. 2: 95–105.
Fee E.J., Hecky R.E., Kasian S.E.M., Cruikshank D.R. 1996: Effects of lake size, water clarity, and climatic variability on mixing depths in Canadian Shield lakes. Limnol. Oceanogr. Vol. 41, No. 5: 912–920.
Ferreira V., Chauvet E. 2011: Synergistic effects of water temperature and dissolved nutrients on litter decomposition and associated fungi. Glob. Change Biol. Vol. 17: 551–564.
Figura S. 2013: The impact of climate change on groundwater temperature and oxygen concentration in Swiss aquifers. ETH Zürich.
Figura S., Livingstone D.M., Kipfer R. 2013: Competing controls on groundwater oxygen concentrations revealed in multidecadal time series from riverbank filtration sites. Water Resour. Res. Vol. 49, No. 11: 7411–7426.
Fink G., Schmid M., Wahl B., Wolf T., Wüest A. 2014: Heat flux modifications related to climate-induced warming of large European lakes. Water Resour. Res. Vol. 50, No. 3: 2072–2085.
Fink G., Wessels M., Wüest A. 2016: Flood frequency matters: Why climate change degrades deep-water quality of peri-alpine lakes. J. Hydrol. Vol. 540: 457–468.
Finn D.S., Khamis K., Milner A.M. 2013: Loss of small glaciers will diminish beta diversity in Pyrenean streams at two levels of biological organization. Glob. Ecol. Biogeogr. Vol. 22, No. 1: 40–51.

Finn D.S., Poff N.L. 2008: Emergence and Flight Activity of Alpine Stream Insects in Two Years with Contrasting Winter Snowpack. Arct. Antarct. Alp. Res. Vol. 40, No. 4: 638–646.
Finn D.S., Räsänen K., Robinson C.T. 2010: Physical and biological changes to a lengthening stream gradient following a decade of rapid glacial recession. Glob. Change Biol. Vol. 16, No. 12: 3314–3326.
Finn D.S., Theobald D.M., Black W.C., Poff N.L. 2006: Spatial population genetic structure and limited dispersal in a Rocky Mountain alpine stream insect. Mol. Ecol. Vol. 15, No. 12: 3553–3566.
Finsinger W., Lotter A.F., Lami A., Kirilova E., Guilizzoni P., Fonville T. 2014: A long-term multi-proxy record of varved sediments suggests climate-induced mixing-regime shift in a large hard-water lake ~5000 years ago. J. Limnol. Vol. 73, No. 2.
Fischer A.M., Keller D.E., Liniger M.A., Rajczak J., Schär C., Appenzeller C. 2015a: Projected changes in precipitation intensity and frequency in Switzerland: a multi-model perspective. Int. J. Climatol. Vol. 35, No. 11: 3204–3219.
Fischer M., Altermatt F., Arlettaz R., Bartha B., Baur B., Bergamini A., Bersier L.-F., Birrer S., Braunisch V., Dollinger P., Eggenberg S., Guisan A., Guntern J., Gutscher H., Herzog F., Humbert J.-Y., Jenny M., Klaus G., Körner C., Krättli H., Küchler M., Lachat T., Lambelet C., Leuzinger Y., Linder P., Mitchell E., Pasinelli G., Pauli D., Pfiffner L., Praz C., Rixen C., Rübel A., Schaffner U., Scheidegger C., Schmid H., Schnyder N., Stöcklin J., Walter T., Zumbach S. 2015b: Zustand der Biodiversität in der Schweiz. Forum Biodiversität Schweiz.
Fischer M., Huss M., Barboux C., Hoelzle M. 2014: The New Swiss Glacier Inventory SGI2010: Relevance of Using High-Resolution Source Data in Areas Dominated by Very Small Glaciers. Arct. Antarct. Alp. Res. Vol. 46, No. 4: 933–945.
Fischer M., Huss M., Hoelzle M. 2015c: Surface elevation and mass changes of all Swiss glaciers 1980–2010. The Cryosphere Vol. 9, No. 2: 525–540.
Fitzhugh R.D., Driscoll C.T., Groffman P.M., Tierney G.L., Fahey T.J., Hardy J.P. 2001: Effects of soil freezing disturbance on soil solution nitrogen, phosphorus, and carbon chemistry in a northern hardwood ecosystem. Biogeochemistry Vol. 56, No. 2: 215–238.
Flory E.A., Milner A.M. 2008: Macroinvertebrate community succession in Wolf Point Creek, Glacier Bay National Park, Alaska. Freshw. Biol. Vol. 44, No. 3: 465–480.
Flury M. 1996: Experimental Evidence of Transport of Pesticides through Field Soils—A Review. J. Environ. Qual. Vol. 25, No. 1: 25–45.
FOEN 2019: Zustand und Entwicklung Grundwasser Schweiz: Ergebnisse der Nationalen Grundwasserbeobachtung NAQUA, Stand 2016. Umwelt-Zustand. Federal Office for the Environment, Bern, Switzerland.
FOEN 2016a: Der Zugersee: Zustand bezüglich Wasserqualität. Federal Office for the Environment, Bern, Switzerland.
FOEN 2016b: Hitze und Trockenheit im Sommer 2015 - Auswirkungen auf Mensch und Umwelt. Umwelt-Zustand. Federal Office for the Environment, Bern, Switzerland.
FOEN 2016c: Genetik und Fischerei. Federal Office for the Environment (FOEN).
FOEN 2013: NAWA – Nationale Beobachtung Oberflächengewässer- qualität. Konzept Fliessgewässer. Federal Office for the Environment (FOEN), Bern, Switzerland.
FOEN 2012: Effects of Climate Change on Water Resources and Waters. Climate Change and Hydrology in Switzerland (CCHydro). Federal Office for the Environment, Bern, Switzerland.
FOEN 2009: Ergebnisse der Grundwasserbeobachtung Schweiz (NAQUA) - Zustand und Entwicklung 2004-2006. Umwelt-Zustand. Federal Office for the Environment, Bern, Switzerland.
FOEN 2007: Methoden zur Untersuchung und Beurteilung der Fliessgewässer: Kieselalgen Stufe F. Federal Office for the Environment (FOEN).
Foley B., Jones I.D., Maberly S.C., Rippey B. 2012: Long-term changes in oxygen depletion in a small temperate lake: effects of climate change and eutrophication. Freshw. Biol. Vol. 57, No. 2: 278–289.

Fortier C., Mailhot A. 2015: Climate Change Impact on Combined Sewer Overflows. J. Water Resour. Plan. Manag. Vol. 141, No. 5: 04014073.
Fourtune L., Paz-Vinas I., Loot G., Prunier J.G., Blanchet S. 2016: Lessons from the fish: a multi-species analysis reveals common processes underlying similar species-genetic diversity correlations. Freshw. Biol. Vol. 61, No. 11: 1830–1845.
Freychet N., Hsu H.-H., Duchez A., Tu C.-Y. 2017: Projection in snowfall characteristics over the European Alps and its sensitivity to the SST changes: results from a 50 km resolution AGCM. Atmospheric Sci. Lett. Vol. 18, No. 6: 261–267.
Friberg N., Dybkjær J.B., Olafsson J.S., Gislason G.M., Larsen S.E., Lauridsen T.L. 2009: Relationships between structure and function in streams contrasting in temperature. Freshw. Biol. Vol. 54, No. 10: 2051–2068.
Frissell C.A., Liss W.J., Warren C.E., Hurley M.D. 1986: A hierarchical framework for stream habitat classification: Viewing streams in a watershed context. Environ. Manage. Vol. 10, No. 2: 199–214.
Fuente M. de la, Bonada N., Bêche L., Dahm C.N., Mendez P.K., Tockner K., Uehlinger U., Acuña V. 2017: Evolutionary responses of aquatic macroinvertebrates to two contrasting flow regimes. Hydrobiologia Vol. 808: 1–18.
Fuhrer J., Beniston M., Fischlin A., Frei C., Goyette S., Jasper K., Pfister C. 2006: Climate risks and their impact on agriculture and forests in Switzerland. Clim. Change Vol. 79: 79–102.
Galil B., Nehring S., Panov V. 2008: Waterways as Invasion Highways – Impact of Climate Change and Globalization. In: Biological InvasionsSpringer, p. 59–74.
Gallardo B., Clavero M., Sánchez M.I., Vilà M. 2016: Global ecological impacts of invasive species in aquatic ecosystems. Glob. Change Biol. Vol. 22, No. 1: 151–163.
Garnier A., Pennekamp F., Lemoine M., Petchey O.L. 2017: Temporal scale dependent interactions between multiple environmental disturbances in microcosm ecosystems. Glob. Change Biol. Vol. 23, No. 12: 5237–5248.
Gaudard A., Weber C., Alexander T.J., Hunziker S., Schmid M. 2018: Impacts of using lakes and rivers for extraction and disposal of heat. Wiley Interdiscip. Rev. Water Vol. 5, No. 5: e1295.
Gleick P.H. 1989: Climate change, hydrology, and water resources. Rev. Geophys. Vol. 27, No. 3: 329–344.
Goode J.R., Buffington J.M., Tonina D., Isaak D.J., Thurow R.F., Wenger S., Nagel D., Luce C., Tetzlaff D., Soulsby C. 2013: Potential effects of climate change on streambed scour and risks to salmonid survival in snow-dominated mountain basins. Hydrol. Process. Vol. 27: 750–765.
Gounand I., Little C.J., Harvey E., Altermatt F. 2018: Cross-ecosystem carbon flows connecting ecosystems worldwide. Nat. Commun. Vol. 9, No. 1: 4825.
Gray L.J., Fisher S.G. 1981: Postflood Recolonization Pathways of Macroinvertebrates in a Lowland Sonoran Desert Stream. Am. Midl. Nat. Vol. 106, No. 2: 249–257.
Green P.A., Vörösmarty C.J., Harrison I., Farrell T., Sáenz L., Fekete B.M. 2015: Freshwater ecosystem services supporting humans: Pivoting from water crisis to water solutions. Glob. Environ. Change Vol. 34: 108–118.
Groffman P.M., Driscoll C.T., Fahey T.J., Hardy J.P., Fitzhugh R.D., Tierney G.L. 2001: Colder soils in a warmer world: A snow manipulation study in a northern hardwood forest ecosystem. Biogeochemistry Vol. 56, No. 2: 135–150.
Gudasz C., Bastviken D., Steger K., Premke K., Sobek S., Tranvik L.J. 2010: Temperature-controlled organic carbon mineralization in lake sediments. Nature Vol. 466, No. 7305: 478.
Gurung A.B., Stähli M. 2014: Wasserressourcen der Schweiz: Dargebot und Nutzung - heute und morgen. Nationales Forschungsprogramm NFP 61 “Nachhaltige Wassernutzung.” Bern.
Haeberli W., Beniston M. 1998: Climate Change and Its Impacts on Glaciers and Permafrost in the Alps. Ambio Vol. 27, No. 4: 258–265.

Haeberli W., Schleiss A., Linsbauer A., Künzler A., Bütler M. 2012: Gletscherschwund und neue Seen in den Schweizer Alpen: Perspektiven und Optionen im Bereich Naturgefahren und Wasserkraft. Wasser Energ. Luft Vol. 104, No. 2: 93–102.
Hagedorn F., Krause H.-M., Studer M., Schellenberger A., Gattinger A. 2018: Boden und Umwelt - Organische Bodensubstanz, Treibhausgasemissionen und physikalische Belastung von Schweizer Böden. Nationales Forschungsprogramm NFP 68 “Nachhaltige Nutzung der Ressource Boden.” Bern.
Hari R.E., Livingstone D.M., Siber R., Burkhardt-Holm P., Güttinger H. 2006: Consequences of climatic change for water temperature and brown trout populations in Alpine rivers and streams. Glob. Change Biol. Vol. 12, No. 1: 10–26.
Hartikainen H., Okamura B. 2015: Ecology and Evolution of Malacosporean-Bryozoan Interactions. In: Myxozoan Evolution, Ecology and Development, p. 201–216. Springer, Cham.
Hauer C., Unfer G., Holzmann H., Schmutz S., Habersack H. 2013: The impact of discharge change on physical instream habitats and its response to river morphology. Clim. Change Vol. 116, No. 3–4: 827–850.
Havens K.E., Sharfstein B., Brady M.A., East T.L., Harwell M.C., Maki R.P., Rodusky A.J. 2004: Recovery of submerged plants from high water stress in a large subtropical lake in Florida, USA. Aquat. Bot. Vol. 78, No. 1: 67–82.
Hedrick R.P., Baxa D.V., Kinkelin P.D., Okamura B. 2004: Malacosporean-like spores in urine of rainbow trout react with antibody and DNA probes to Tetracapsuloides bryosalmonae. Parasitol. Res. Vol. 92, No. 1: 81–88.
Hedrick R.P., MacConnell E., de Kinkelin P. 1993: Proliferative kidney disease of salmonid fish. Annu. Rev. Fish Dis. Vol. 3: 277–290.
Heinz B., Birk S., Liedl R., Geyer T., Straub K.L., Andresen J., Bester K., Kappler A. 2009: Water quality deterioration at a karst spring (Gallusquelle, Germany) due to combined sewer overflow: evidence of bacterial and micro-pollutant contamination. Environ. Geol. Vol. 57, No. 4: 797–808.
Hejzlar J., Dubrovský M., Buchtele J., Růžička M. 2003: The apparent and potential effects of climate change on the inferred concentration of dissolved organic matter in a temperate stream (the Malše River, South Bohemia). Sci. Total Environ., Detecting Environmental Change: Science and Society Vol. 310, No. 1: 143–152.
Hellmann J.J., Byers J.E., Bierwagen B.G., Dukes J.S. 2008: Five Potential Consequences of Climate Change for Invasive Species. Conserv. Biol. Vol. 22, No. 3: 534–543.
Herb W.R., Janke B., Mohseni O., Stefan H.G. 2008: Thermal pollution of streams by runoff from paved surfaces. Hydrol. Process. Vol. 22, No. 7: 987–999.
Hershkovitz Y., Dahm V., Lorenz A.W., Hering D. 2015: A multi-trait approach for the identification and protection of European freshwater species that are potentially vulnerable to the impacts of climate change. Ecol. Indic. Vol. 50, No. Supplement C: 150–160.
Hilt S., Köhler J., Kozerski H.-P., Nes E.H. van, Scheffer M. 2011: Abrupt regime shifts in space and time along rivers and connected lake systems. Oikos Vol. 120, No. 5: 766–775.
Hiltbrunner E., Schwikowski M., Körner C. 2005: Inorganic N storage in alpine snow pack in the Central Alps (Switzerland). Atmos. Environ. Vol. 39: 2249–2259.
Hoekman D. 2010: Turning up the heat: Temperature influences the relative importance of top-down and bottom-up effects. Ecology Vol. 91, No. 10: 2819–2825.
Hoffmann S., Hunkeler D., Maurer M. 2014: Nachhaltige Wasserversorgung und Abwasserentsorgung in der Schweiz: Herausforderungen und Handlungsoptionen. Nationales Forschungsprogramm NFP 61 “Nachhaltige Wassernutzung.” Bern.
Honti M., Schuwirth N., Rieckermann J., Stamm C. 2017: Can integrative catchment management mitigate future water quality issues caused by climate change and socio-economic development? Hydrol Earth Syst Sci Vol. 21, No. 3: 1593–1609.
Hotaling S., Finn D.S., Joseph Giersch J., Weisrock D.W., Jacobsen D. 2017: Climate change and alpine stream biology: progress, challenges, and opportunities for the future. Biol. Rev. Vol. 92, No. 4: 2024–2045.

Hruška J., Krám P., McDowell W.H., Oulehle F. 2009: Increased Dissolved Organic Carbon (DOC) in Central European Streams is Driven by Reductions in Ionic Strength Rather than Climate Change or Decreasing Acidity. Environ. Sci. Technol. Vol. 43, No. 12: 4320–4326.
Hughes A.R., Inouye B.D., Johnson M.T.J., Underwood N., Vellend M. 2008: Ecological consequences of genetic diversity. Ecol. Lett. Vol. 11, No. 6: 609–623.
Hunkeler D., Goldscheider N., Rossi P., Burn C. 2006: Biozönosen im Grundwasser - Grundlagen und Methoden der Charakterisierung von mikrobiellen Gemeinschaften. Umwelt-Wissen. Federal Office for the Environment (FOEN), Bern, Switzerland.
Huntscha S., Stravs M.A., Bühlmann A., Ahrens C.H., Frey J.E., Pomati F., Hollender J., Buerge I.J., Balmer M.E., Poiger T. 2018: Seasonal Dynamics of Glyphosate and AMPA in Lake Greifensee: Rapid Microbial Degradation in the Epilimnion During Summer. Environ. Sci. Technol. Vol. 52, No. 8: 4641–4649.
Huss M. 2011: Present and future contribution of glacier storage change to runoff from macroscale drainage basins in Europe. Water Resour. Res. Vol. 47, No. 7: W07511.
Huss M., Fischer M. 2016: Sensitivity of Very Small Glaciers in the Swiss Alps to Future Climate Change. Front. Earth Sci. Vol. 4: 34.
Ilg C., Castella E. 2006: Patterns of macroinvertebrate traits along three glacial stream continuums. Freshw. Biol. Vol. 51, No. 5: 840–853.
IPCC 2014: Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Core Writing Team, R.K. Pachauri and L.A. Meyer (eds.)]. IPCC, Geneva, Switzerland.
IPCC 2007: Climate Change 2007: The Physical Science Basis. Cambridge Press, United Kingdom.
Isaak D.J., Rieman B.E. 2013: Stream isotherm shifts from climate change and implications for distributions of ectothermic organisms. Glob. Change Biol. Vol. 19, No. 3: 742–751.
Isles P.D.F., Xu Y., Stockwell J.D., Schroth A.W. 2017: Climate-driven changes in energy and mass inputs systematically alter nutrient concentration and stoichiometry in deep and shallow regions of Lake Champlain. Biogeochemistry Vol. 133, No. 2: 201–217.
Jacobsen D., Cauvy-Fraunie S., Andino P., Espinosa R., Cueva D., Dangles O. 2014: Runoff and the longitudinal distribution of macroinvertebrates in a glacier-fed stream: implications for the effects of global warming. Freshw. Biol. Vol. 59, No. 10: 2038–2050.
Jacobsen D., Milner A.M., Brown L.E., Dangles O. 2012: Biodiversity under threat in glacier-fed river systems. Nat. Clim. Change Vol. 2, No. 5: 361–364.
Jeannin P.-Y., Hessenauer M., Malard A., Chapuis V. 2016: Impact of global change on karst groundwater mineralization in the Jura Mountains. Sci. Total Environ. Vol. 541: 1208–1221.
Jenkins A., Ferrier R., Waters D. 1993: Melt water chemistry and its impact on stream water quality. Hydrol. Process. Vol. 7, No. 2: 193–203.
Jeppesen E., Kronvang B., Meerhoff M., Søndergaard M., Hansen K.M., Andersen H.E., Lauridsen T.L., Liboriussen L., Beklioglu M., Özen A., Olesen J.E. 2009: Climate change effects on runoff, catchment phosphorus loading and lake ecological state, and potential adaptations. J. Environ. Qual. Vol. 38, No. 5: 1930–1941.
Jöhnk K.D., Huisman J., Sharples J., Sommeijer B., Visser P.M., Stroom J.M. 2008: Summer heatwaves promote blooms of harmful cyanobacteria. Glob. Change Biol. Vol. 14, No. 3: 495–512.
Johnston C.A., Groffman P., Breshears D.D., Cardon Z.G., Currie W., Emanuel W., Gaudinski J., Jackson R.B., Lajtha K., Nadelhoffer K. 2004: Carbon cycling in soil. Front. Ecol. Environ. Vol. 2, No. 10: 522–528.
Jourdan J., O'Hara R.B., Bottarin R., Huttunen K.-L., Kuemmerlen M., Monteith D., Muotka T., Ozoliņš D., Paavola R., Pilotto F., Springe G., Skuja A., Sundermann A., Tonkin J.D., Haase P. 2018: Effects of changing climate on European stream invertebrate communities: A long-term data analysis. Sci. Total Environ. Vol. 621: 588–599.

Jouvet G., Huss M., Funk M., Blatter H. 2011: Modelling the retreat of Grosser Aletschgletscher, Switzerland, in a changing climate. J. Glaciol. Vol. 57, No. 206: 1033–1045.
Junker J., Heimann F.U.M., Hauer C., Turowski J.M., Rickenmann D., Zappa M., Peter A. 2015: Assessing the impact of climate change on brown trout (Salmo trutta fario) recruitment. Hydrobiologia Vol. 751, No. 1: 1–21.
Kaelin K., Altermatt F. 2016: Landscape-level predictions of diversity in river networks reveal opposing patterns for different groups of macroinvertebrates. Aquat. Ecol. Vol. 50, No. 2: 283–295.
Kalbitz K., Solinger S., Park J.-H., Michalzik B., Matzner E. 2000: Controls on the dynamics of dissolved organic matter: A review. Soil Sci. Vol. 165, No. 4: 277.
Kallenborn R., Halsall C., Dellong M., Carlsson P. 2012: The influence of climate change on the global distribution and fate processes of anthropogenic persistent organic pollutants. J. Environ. Monit. JEM Vol. 14, No. 11: 2854–2869.
Kipfer R., Livingstone D.M. 2008: Water resources and climate change. Eawag, Dübendorf, Switzerland.
Kirschbaum M.U.F. 1995: The temperature dependence of soil organic matter decomposition, and the effect of global warming on soil organic C storage. Soil Biol. Biochem. Vol. 27, No. 6: 753–760.
Klausmeier C.A., Litchman E. 2001: Algal games: The vertical distribution of phytoplankton in poorly mixed water columns. Limnol. Oceanogr. Vol. 46, No. 8: 1998–2007.
Klein G., Vitasse Y., Rixen C., Marty C., Rebetez M. 2016: Shorter snow cover duration since 1970 in the Swiss Alps due to earlier snowmelt more than to later snow onset. Clim. Change Vol. 139, No. 3–4: 637–649.
KLIWA 2015: KLIWA - Einfluss des Klimawandels auf Seen - Literaturauswertungsstudie. Kooperationsvorhaben Klimaveränderung und Wasserwirtschaft (KLIWA), Konstanz.
Koinig K.A., Schmidt R., Sommaruga-Wögrath S., Tessadri R., Psenner R. 1998: Climate Change as the Primary Cause for pH Shifts in a High Alpine Lake. Water. Air. Soil Pollut. Vol. 104, No. 1–2: 167–180.
Köplin N., Schädler B., Viviroli D., Weingartner R. 2014: Seasonality and magnitude of floods in Switzerland under future climate change. Hydrol. Process. Vol. 28, No. 4: 2567–2578.
Kötzsch S., Sinreich M. 2014: Zellzahlen zum Grundwasser – Bestimmung mittels Durchflusszytometrie. Aqua Gas No. 3: 14–21.
Kritzberg E.S., Granéli W., Björk J., Brönmark C., Hallgren P., Nicolle A., Persson A., Hansson L.-A. 2014: Warming and browning of lakes: consequences for pelagic carbon metabolism and sediment delivery. Freshw. Biol. Vol. 59, No. 2: 325–336.
Kunz M., Schindler Wildhaber Y., Dietzel A. 2016: Zustand der Schweizer Fliessgewässer. Ergebnisse der Nationalen Beobachtung Oberflächengewässerqualität (NAWA) 2011–2014, Bern.
Lair G.J., Zehetner F., Fiebig M., Gerzabek M.H., van Gestel C.A.M., Hein T., Hohensinner S., Hsu P., Jones K.C., Jordan G., Koelmans A.A., Poot A., Slijkerman D.M.E., Totsche K.U., Bondar-Kunze E., Barth J.A.C. 2009: How do long-term development and periodical changes of river–floodplain systems affect the fate of contaminants? Results from European rivers. Environ. Pollut., Persistent Organic Pollutants in Mountainous Areas Vol. 157, No. 12: 3336–3346.
Lanz K., Rahn E., Siber R., Stamm C. 2014: Bewirtschaftung der Wasserressourcen unter steigendem Nutzungsdruck. Nationales Forschungsprogramm NFP 61 "Nachhaltige Wassernutzung." Bern.
Lencioni V. 2018: Glacial influence and stream macroinvertebrate biodiversity under climate change: Lessons from the Southern Alps. Sci. Total Environ. Vol. 622–623: 563–575.
Lencioni V., Bernabò P. 2017: Thermal survival limits of young and mature larvae of a cold stenothermal chironomid from the Alps (Diamesinae: Pseudodiamesa branickii [Nowicki, 1873]). Insect Sci. Vol. 24, No. 2: 314–324.
Lepori F., Pozzoni M., Pera S. 2015: What Drives Warming Trends in Streams? A Case Study from the Alpine Foothills. River Res. Appl. Vol. 31, No. 6: 663–675.

Lepori F., Roberts J.J. 2015: Past and future warming of a deep European lake (Lake Lugano): What are the climatic drivers? J. Gt. Lakes Res. Vol. 41, No. 4: 973–981.

Leuven R.S.E.W., van der Velde G., Baijens I., Snijders J., van der Zwart C., Lenders H.J.R., bij de Vaate A. 2009: The river Rhine: a global highway for dispersal of aquatic invasive species. Biol. Invasions Vol. 11, No. 9: 1989–2008.

Liboriussen L., Lauridsen T.L., Søndergaard M., Landkildehus F., Søndergaard M., Larsen S.E., Jeppesen E. 2011: Effects of warming and nutrients on sediment community respiration in shallow lakes: an outdoor mesocosm experiment. Freshw. Biol. Vol. 56, No. 3: 437–447.

Lin B.-L., Sakoda A., Shibasaki R., Suzuki M. 2001: A Modelling Approach to Global Nitrate Leaching Caused by Anthropogenic Fertilisation. Water Res. Vol. 35, No. 8: 1961–1968.

Lind L., Nilsson C., Polvi L.E., Weber C. 2014: The role of ice dynamics in shaping vegetation in flowing waters. Biol. Rev. Vol. 89, No. 4: 791–804.

Lindner M., Maroschek M., Netherer S., Kremer A., Barbati A., Garcia-Gonzalo J., Seidl R., Delzon S., Corona P., Kolström M., Lexer M.J., Marchetti M. 2010: Climate change impacts, adaptive capacity, and vulnerability of European forest ecosystems. For. Ecol. Manag., Adaptation of Forests and Forest Management to Changing Climate Vol. 259, No. 4: 698–709.

Linsbauer A., Paul F., Machguth H., Haeberli W. 2013: Comparing three different methods to model scenarios of future glacier change in the Swiss Alps. Ann. Glaciol. Vol. 54, No. 63: 241–253.

Little C.J., Altermatt F. 2018: Species turnover and invasion of dominant freshwater invertebrates alter biodiversity–ecosystem-function relationship. Ecol. Monogr. Vol. 88, No. 3: 461–480.

Longhi M.L., Beisner B.E. 2009: Environmental factors controlling the vertical distribution of phytoplankton in lakes. J. Plankton Res. Vol. 31, No. 10: 1195–1207.

Loreau M., de Mazancourt C. 2013: Biodiversity and ecosystem stability: a synthesis of underlying mechanisms. Ecol. Lett. Vol. 16 Suppl 1: 106–115.

Lytle D.A. 2002: Flash Floods and Aquatic Insect Life-History Evolution: Evaluation of Multiple Models. Ecology Vol. 83, No. 2: 370–385.

Lytle D.A., Bogan M.T., Finn D.S. 2008: Evolution of aquatic insect behaviours across a gradient of disturbance predictability. Proc. R. Soc. Lond. B Biol. Sci. Vol. 275, No. 1633: 453–462.

Lytle D.A., Poff N.L. 2004: Adaptation to natural flow regimes. Trends Ecol. Evol. Vol. 19, No. 2: 94–100.

Magnuson J.J., Robertson D.M., Benson B.J., Wynne R.H., Livingstone D.M., Arai T., Assel R.A., Barry R.G., Card V., Kuusisto E., Granin N.G., Prowse T.D., Stewart K.M., Vuglinski V.S. 2000: Historical trends in lake and river ice cover in the Northern Hemisphere. Science Vol. 289, No. 5485: 1743–1746.

Magnuson J.J., Webster K.E., Assel R.A., Bowser C.J., Dillon P.J., Eaton J.G., Evans H.E., Fee E.J., Hall R.I., Mortsch L.R., Schindler D.W., Quinn F.H. 1997: Potential Effects of Climate Changes on Aquatic Systems: Laurentian Great Lakes and Precambrian Shield Region. Hydrol. Process. Vol. 11, No. 8: 825–871.

Malard F., Galassi D., Lafont M., Dolédec S., Ward J. v. 2003: Longitudinal patterns of invertebrates in the hyporheic zone of a glacial river. Freshw. Biol. Vol. 48, No. 10: 1709–1725.

Marty C., Schlögl S., Bavay M., Lehning M. 2017: How much can we save? Impact of different emission scenarios on future snow cover in the Alps. The Cryosphere Vol. 11, No. 1: 517–529.

Matzner E., Borken W. 2008: Do freeze-thaw events enhance C and N losses from soils of different ecosystems? A review. Eur. J. Soil Sci. Vol. 59, No. 2: 274–284.

Mccauley S.J., Hammond J.I., Frances D.N., Mabry K.E. 2015: Effects of experimental warming on survival, phenology, and morphology of an aquatic insect (Odonata). Ecol. Entomol. Vol. 40, No. 3: 211–220.

Meier W., Bonjour C., Wüest A., Reichert P. 2003: Modeling the Effect of Water Diversion on the Temperature of Mountain Streams. J. Environ. Eng. Vol. 129, No. 8: 755–764.
Menéndez R., Megías A.G., Hill J.K., Braschler B., Willis S.G., Collingham Y., Fox R., Roy D.B., Thomas C.D. 2006: Species richness changes lag behind climate change. Proc. R. Soc. Lond. B Biol. Sci. Vol. 273, No. 1593: 1465–1470.
MeteoSwiss 2018: Climate Change in Switzerland. WWW Document.
MFR, ISSKA 2012: Analyse et interprétation des tendances physico-chimiques enregistrées dans le cadre de la surveillance qualitative des points d'eau du réseau hydrométrique de l'A16 (1989-2011).
Milly P.C.D., Wetherald R.T., Dunne K.A., Delworth T.L. 2002: Increasing risk of great floods in a changing climate. Nature Vol. 415, No. 6871: 514–517.
Milner A.M., Brittain J.E., Castella E., Petts G.E. 2001: Trends of macroinvertebrate community structure in glacier-fed rivers in relation to environmental conditions: a synthesis. Freshw. Biol. Vol. 46, No. 12: 1833–1847.
Milner A.M., Brown L.E., Hannah D.M. 2009: Hydroecological response of river systems to shrinking glaciers. Hydrol. Process. Vol. 23, No. 1: 62–77.
Minnig M., Moeck C., Radny D., Schirmer M. 2017: Impact of urbanization on groundwater recharge rates in Dübendorf, Switzerland. J. Hydrol. Vol. 563: 1135–1146.
Mohseni O., Stefan H.G. 1999: Stream temperature/air temperature relationship: a physical interpretation. J. Hydrol. Vol. 218, No. 3: 128–141.
Monchamp M.-E., Spaak P., Domaizon I., Dubois N., Bouffard D., Pomati F. 2017: Homogenization of lake cyanobacterial communities over a century of climate change and eutrophication. Nat. Ecol. Evol. Vol. 2: 317–324.
Monteith D.T., Stoddard J.L., Evans C.D., de Wit H.A., Forsius M., Høgåsen T., Wilander A., Skjelkvåle B.L., Jeffries D.S., Vuorenmaa J., Keller B., Kopácek J., Vesely J. 2007: Dissolved organic carbon trends resulting from changes in atmospheric deposition chemistry. Nature Vol. 450, No. 7169: 537–540.
Moore M.V., Pace M.L., Mather J.R., Murdoch P.S., Howarth R.W., Folt C.L., Chen C.Y., Hemond H.F., Flebbe P.A., Driscoll C.T. 1997: Potential effects of climate change on freshwater ecosystems of the New England/Mid-Atlantic Region. Hydrol. Process. Vol. 11, No. 8: 925–947.
Morán-Tejeda E., López-Moreno J.I., Stoffel M., Beniston M. 2016: Rain-on-snow events in Switzerland: recent observations and projections for the 21st century. Clim. Res. Vol. 71, No. 2: 111–125.
Mulholland P.J., Best G.R., Coutant C.C., Hornberger G.M., Meyer J.L., Robinson P.J., Stenberg J.R., Turner R.E., Vera-Herrera F., Wetzel R.G. 1997: Effects of Climate Change on Freshwater Ecosystems of the South-Eastern United States and the Gulf Coast of Mexico. Hydrol. Process. Vol. 11, No. 8: 949–970.
Müller B., Och L., Wüest A. 2012: Entwicklung des Phosphorhaushalts und der Sauerstoffzehrung im Sempacher- und Baldeggersee. Kastanienbaum.
Murdoch P.S., Baron J.S., Miller T.L. 2000: Potential Effects of Climate Change on Surface-Water Quality in North America. JAWRA J. Am. Water Resour. Assoc. Vol. 36, No. 2: 347–366.
Musselman K.N., Clark M.P., Liu C., Ikeda K., Rasmussen R. 2017: Slower snowmelt in a warmer world. Nat. Clim. Change Vol. 7, No. 3: 214.
Nelson D., Benstead J.P., Huryn A.D., Cross W.F., Hood J.M., Johnson P.W., Junker J.R., Gislason G.M., Olafsson J.S. 2017a: Experimental whole-stream warming alters community size structure. Glob. Change Biol. Vol. 23, No. 7: 2618–2628.
Nelson D., Benstead J.P., Huryn A.D., Cross W.F., Hood J.M., Johnson P.W., Junker J.R., Gíslason G.M., Ólafsson J.S. 2017b: Shifts in community size structure drive temperature invariance of secondary production in a stream-warming experiment. Ecology Vol. 98, No. 7: 1797–1806.
Nelson K.C., Palmer M.A. 2007: Stream temperature surges under urbanization and climate change: Data, models, and responses. J. Am. Water Resour. Assoc. Vol. 43, No. 2: 440–452.

Nie L., Lindholm O., Lindholm G., Syversen E. 2009: Impacts of climate change on urban drainage systems – a case study in Fredrikstad, Norway. Urban Water J. Vol. 6, No. 4: 323–332.
North R.P., Livingstone D.M., Hari R.E., Köster O., Niederhauser P., Kipfer R. 2013: The physical impact of the late 1980s climate regime shift on Swiss rivers and lakes. Inland Waters Vol. 3, No. 3: 341–350.
Noyes P.D., McElwee M.K., Miller H.D., Clark B.W., Van Tiem L.A., Walcott K.C., Erwin K.N., Levin E.D. 2009: The toxicology of climate change: Environmental contaminants in a warming world. Environ. Int. Vol. 35, No. 6: 971–986.
O'Gorman E.J., Ólafsson Ó.P., Demars B.O.L., Friberg N., Guðbergsson G., Hannesdóttir E.R., Jackson M.C., Johansson L.S., McLaughlin Ó.B., Ólafsson J.S., Woodward G., Gíslason G.M. 2016: Temperature effects on fish production across a natural thermal gradient. Glob. Change Biol. Vol. 22, No. 9: 3206–3220.
O'Gorman E.J., Zhao L., Pichler D.E., Adams G., Friberg N., Rall B.C., Seeney A., Zhang H., Reuman D.C., Woodward G. 2017: Unexpected changes in community size structure in a natural warming experiment. Nat. Clim. Change Vol. 7, No. 9: 659–663.
O'Reilly C.M., Sharma S., Gray D.K., Hampton S.E., Read J.S., Rowley R.J., Schneider P., Lenters J.D., McIntyre P.B., Kraemer B.M., Weyhenmeyer G.A., Straile D., Dong B., Adrian R., Allan M.G., Anneville O., Arvola L., Austin J., Bailey J.L., Baron J.S., Brookes J.D., de Eyto E., Dokulil M.T., Hamilton D.P., Havens K., Hetherington A.L., Higgins S.N., Hook S., Izmest'eva L.R., Joehnk K.D., Kangur K., Kasprzak P., Kumagai M., Kuusisto E., Leshkevich G., Livingstone D.M., MacIntyre S., May L., Melack J.M., Mueller-Navarra D.C., Naumenko M., Noges P., Noges T., North R.P., Plisnier P.-D., Rigosi A., Rimmer A., Rogora M., Rudstam L.G., Rusak J.A., Salmaso N., Samal N.R., Schindler D.E., Schladow S.G., Schmid M., Schmidt S.R., Silow E., Soylu M.E., Teubner K., Verburg P., Voutilainen A., Watkinson A., Williamson C.E., Zhang G. 2015: Rapid and highly variable warming of lake surface waters around the globe. Geophys. Res. Lett. Vol. 42, No. 24: 2015GL066235.
Özen A., Šorf M., Trochine C., Liboriussen L., Beklioglu M., Søndergaard M., Lauridsen T.L., Johansson L.S., Jeppesen E. 2013: Long-term effects of warming and nutrients on microbes and other plankton in mesocosms. Freshw. Biol. Vol. 58, No. 3: 483–493.
Paerl H.W. 2014: Mitigating Harmful Cyanobacterial Blooms in a Human- and Climatically-Impacted World. Life Vol. 4, No. 4: 988–1012.
Paerl H.W., Huisman J. 2008: Blooms Like It Hot. Science Vol. 320, No. 5872: 57–58.
Paillex A., Castella E., zu Ermgassen P., Gallardo B., Aldridge D.C. 2017: Large river floodplain as a natural laboratory: non-native macroinvertebrates benefit from elevated temperatures. Ecosphere Vol. 8, No. 10: e01972.
Pandit S.N., Maitland B.M., Pandit L.K., Poesch M.S., Enders E.C. 2017: Climate change risks, extinction debt, and conservation implications for a threatened freshwater fish: Carmine shiner (Notropis percobromus). Sci. Total Environ. Vol. 598, No. Supplement C: 1–11.
Patz J.A., Vavrus S.J., Uejio C.K., McLellan S.L. 2008: Climate Change and Waterborne Disease Risk in the Great Lakes Region of the U.S. Am. J. Prev. Med., Theme Issue: Climate Change and the Health of the Public Vol. 35, No. 5: 451–458.
Pavoine S., Marcon E., Ricotta C. 2016: 'Equivalent numbers' for species, phylogenetic or functional diversity in a nested hierarchy of multiple scales. Methods Ecol. Evol. Vol. 7, No. 10: 1152–1163.
Paz-Vinas I., Loot G., Stevens V.M., Blanchet S. 2015: Evolutionary processes driving spatial patterns of intraspecific genetic diversity in river ecosystems. Mol. Ecol. Vol. 24, No. 18: 4586–4604.
Peeters F., Livingstone D.M., Goudsmit G.-H., Kipfer R., Forster R. 2002: Modeling 50 years of historical temperature profiles in a large central European lake. Limnol. Oceanogr. Vol. 47, No. 1: 186–197.
Pellicciotti F., Carenzo M., Bordoy R., Stoffel M. 2014: Changes in glaciers in the Swiss Alps and impact on basin hydrology: Current state of the art and future research. Sci. Total Environ. Vol. 493, No. Supplement C: 1152–1170.

Perga M.-E., Bruel R., Rodriguez L., Guénand Y., Bouffard D. 2018: Storm impacts on alpine lakes: Antecedent weather conditions matter more than the event intensity. Glob. Change Biol. Vol. 24, No. 10: 5004–5016.

Perroud M., Bader S. 2013: Klimaänderung in der Schweiz. Indikatoren zu Ursachen, Auswirkungen, Massnahmen. Umwelt-Zustand. Federal Office for the Environment (FOEN) and Federal Office of Meteorology and Climatology (MeteoSwiss).

Person E., Bieri M., Peter A., Schleiss A.J. 2014: Mitigation measures for fish habitat improvement in Alpine rivers affected by hydropower operations. Ecohydrology Vol. 7, No. 2: 580–599.

Petchey O.L., Downing A.L., Mittelbach Gary G., Persson Lennart, Steiner Christopher F., Warren Philip H., Woodward Guy 2004: Species loss and the structure and functioning of multitrophic aquatic systems. Oikos Vol. 104, No. 3: 467–478.

Petchey O.L., McPhearson P.T., Casey T.M., Morin P.J. 1999: Environmental warming alters food-web structure and ecosystem function. Nature Vol. 402, No. 6757: 69–72.

Poff N.L., Larson E.I., Salerno P.E., Morton S.G., Kondratieff B.C., Flecker A.S., Zamudio K.R., Funk W.C. 2018: Extreme streams: species persistence and genomic change in montane insect populations across a flooding gradient. Ecol. Lett. Vol. 21, No. 4: 525–535.

Poff N.L., Olden J.D., Vieira N.K.M., Finn D.S., Simmons M.P., Kondratieff B.C. 2006: Functional trait niches of North American lotic insects: traits-based ecological applications in light of phylogenetic relationships. J. North Am. Benthol. Soc. Vol. 25, No. 4: 730–755.

Pomati F., Matthews B., Seehausen O., Ibelings B.W. 2017: Eutrophication and climate warming alter spatial (depth) co-occurrence patterns of lake phytoplankton assemblages. Hydrobiologia Vol. 787, No. 1: 375–385.

Pompini M., Buser A.M., Thali M.R., Von Siebenthal B.A., Nusslé S., Guduff S., Wedekind C. 2013: Temperature-induced sex reversal is not responsible for sex-ratio distortions in grayling Thymallus thymallus or brown trout Salmo trutta. J. Fish Biol. Vol. 83, No. 2: 404–411.

Porter K.G., Saunders P.A., Haberyan K.A., Macubbin A.E., Jacobsen T.R., Hodson R.E. 1996: Annual cycle of autotrophic and heterotrophic production in a small, monomictic Piedmont lake (Lake Oglethorpe): Analog for the effects of climatic warming on dimictic lakes. Limnol. Oceanogr. Vol. 41, No. 5: 1041–1051.

Posch T., Köster O., Salcher M.M., Pernthaler J. 2012: Harmful filamentous cyanobacteria favoured by reduced water turnover with lake warming. Nat. Clim. Change Vol. 2, No. 11: nclimate1581.

Prasuhn V. 2012: On-farm effects of tillage and crops on soil erosion measured over 10 years in Switzerland. Soil Tillage Res. Vol. 120: 137–146.

Prasuhn V., Albisser C.V. 2014: Grundwasserqualität und Bewässerung - Eine Lysimeterstudie zur Schadstoffverfrachtung ins Grundwasser. Aqua Gas No. 4: 54–58.

Rahel F.J., Bierwagen B., Taniguchi Y. 2008: Managing Aquatic Species of Conservation Concern in the Face of Climate Change and Invasive Species. Conserv. Biol. Vol. 22, No. 3: 551–561.

Rahel F.J., Olden J.D. 2008: Assessing the Effects of Climate Change on Aquatic Invasive Species. Conserv. Biol. Vol. 22, No. 3: 521–533.

Rahman K., Besacier-Monbertrand A.-L., Castella E., Lods-Crozet B., Ilg C., Beguin O. 2015: Quantification of the daily dynamics of streamflow components in a small alpine watershed in Switzerland using end member mixing analysis. Environ. Earth Sci. Vol. 74, No. 6: 4927–4937.

Rajczak J., Pall P., Schär C. 2013: Projections of extreme precipitation events in regional climate simulations for Europe and the Alpine Region. J. Geophys. Res. Atmospheres Vol. 118, No. 9: 3610–3626.

Råman Vinnå L., Wüest A., Zappa M., Fink G., Bouffard D. 2018: Tributaries affect the thermal response of lakes to climate change. Hydrol Earth Syst Sci Vol. 22, No. 1: 31–51.

Rasconi S., Gall A., Winter K., Kainz M.J. 2015: Increasing Water Temperature Triggers Dominance of Small Freshwater Plankton. PLOS ONE Vol. 10, No. 10: e0140449.

Rebetez M., Reinhard M. 2008: Monthly air temperature trends in Switzerland 1901–2000 and 1975–2004. Theor. Appl. Climatol. Vol. 91, No. 1–4: 27–34.

Robinson C.T., Gessner M.O. 2000: Nutrient addition accelerates leaf breakdown in an alpine springbrook. Oecologia Vol. 122, No. 2: 258–263.

Robinson C.T., Thompson C., Freestone M. 2014: Ecosystem development of streams lengthened by rapid glacial recession. Fundam. Appl. Limnol. Arch. Fr Hydrobiol. Vol. 185, No. 3–4: 235–246.

Rodríguez-Murillo J.C., Filella M. 2015: Temporal evolution of organic carbon concentrations in Swiss lakes: Trends of allochthonous and autochthonous organic carbon. Sci. Total Environ. Vol. 520: 13–22.

Rodríguez-Murillo J.C., Zobrist J., Filella M. 2015: Temporal trends in organic carbon content in the main Swiss rivers, 1974–2010. Sci. Total Environ. Vol. 502: 206–217.

Rolls R.J., Heino J., Ryder D.S., Chessman B.C., Growns I.O., Thompson R.M., Gido K.B. 2017: Scaling biodiversity responses to hydrological regimes. Biol. Rev. Vol. 93, No. 2: 971–995.

Rosset V., Oertli B. 2011: Freshwater biodiversity under climate warming pressure: Identifying the winners and losers in temperate standing waterbodies. Biol. Conserv. Vol. 144, No. 9: 2311–2319.

Rossi L., Chèvre N., Fankhauser R., Margot J., Curdy R., Babut M., Barry D.A. 2013: Sediment contamination assessment in urban areas based on total suspended solids. Water Res. Vol. 47, No. 1: 339–350.

Rossi L., de Alencastro L., Kupper T., Tarradellas J. 2004: Urban stormwater contamination by polychlorinated biphenyls (PCBs) and its importance for urban water systems in Switzerland. Sci. Total Environ. Vol. 322, No. 1: 179–189.

Rossi L., Hari R.E. 2009: Screening procedure to assess the impact of urban stormwater temperature to populations of brown trout in receiving water. Integr. Environ. Assess. Manag. Vol. 3, No. 3: 383–392.

Rouse W.R., Douglas M.S.V., Hecky R.E., Hershey A.E., Kling G.W., Lesack L., Marsh P., Mcdonald M., Nicholson B.J., Roulet N.T., Smol J.P. 1997: Effects of Climate Change on the Freshwaters of Arctic and Subarctic North America. Hydrol. Process. Vol. 11, No. 8: 873–902.

Royer T.V., David M.B., Gentry L.E. 2006: Timing of Riverine Export of Nitrate and Phosphorus from Agricultural Watersheds in Illinois: Implications for Reducing Nutrient Loading to the Mississippi River. Environ. Sci. Technol. Vol. 40, No. 13: 4126–4131.

Rüegg J., Robinson C.T. 2004: Comparison of macroinvertebrate assemblages of permanent and temporary streams in an Alpine flood plain, Switzerland. Arch. Für Hydrobiol. Vol. 161, No. 4: 489–510.

Sadro S., Sickman J.O., Melack J.M., Skeen K. 2018: Effects of Climate Variability on Snowmelt and Implications for Organic Matter in a High-Elevation Lake. Water Resour. Res. Vol. 54, No. 7: 4563–4578.

Salmaso N., Boscaini A., Capelli C., Cerasino L. 2017: Ongoing ecological shifts in a large lake are driven by climate change and eutrophication: evidences from a three-decade study in Lake Garda. Hydrobiologia: 1–19.

Salmaso N., Capelli C., Shams S., Cerasino L. 2015: Expansion of bloom-forming Dolichospermum lemmermannii (Nostocales, Cyanobacteria) to the deep lakes south of the Alps: Colonization patterns, driving forces and implications for water use. Harmful Algae Vol. 50: 76–87.

Salzmann N., Machguth H., Linsbauer A. 2012: The Swiss Alpine glaciers' response to the global '2 °C air temperature target.' Environ. Res. Lett. Vol. 7, No. 4: 044001.

Schaffner M., Studer P., Ramseier C. 2013: Beurteilung der Badegewässer - Empfehlungen zur Untersuchung und Beurteilung der Badewasserqualität von See- und Flussbädern. Umwelt-Vollzug. Federal Office for the Environment (FOEN), Bern, Switzerland.

Scheffer M. 2004: Ecology of Shallow Lakes. Population and Community Biology Series. Springer Netherlands.
Scheffer M., Carpenter S., Foley J.A., Folke C., Walker B. 2001: Catastrophic shifts in ecosystems. Nature Vol. 413, No. 6856: 591–596.
Scheffer M., Hosper S.H., Meijer M.-L., Moss B., Jeppesen E. 1993: Alternative equilibria in shallow lakes. Trends Ecol. Evol. Vol. 8, No. 8: 275–279.
Scherrer S.C., Fischer E.M., Posselt R., Liniger M.A., Croci-Maspoli M., Knutti R. 2016: Emerging trends in heavy precipitation and hot temperature extremes in Switzerland. J. Geophys. Res. Atmospheres Vol. 121, No. 6: 2015JD024634.
Scheurer K., Alewell C., Bänninger D., Burkhardt-Holm P. 2009: Climate and land-use changes affecting river sediment and brown trout in alpine countries--a review. Environ. Sci. Pollut. Res. Int. Vol. 16, No. 2: 232–242.
Schimel J., Balser T.C., Wallenstein M. 2007: Microbial stress-response physiology and its implications for ecosystem function. Ecology Vol. 88, No. 6: 1386–1394.
Schindler D.W. 2009: Lakes as sentinels and integrators for the effects of climate change on watersheds, airsheds, and landscapes. Limnol. Oceanogr. Vol. 54, No. 6, part 2: 2349–2358.
Schindler D.W. 1997: Widespread Effects of Climatic Warming on Freshwater Ecosystems in North America. Hydrol. Process. Vol. 11, No. 8: 1043–1067.
Schlesinger W.H., Dietze M.C., Jackson R.B., Phillips R.P., Rhoades C.C., Rustad L.E., Vose J.M. 2015: Forest biogeochemistry in response to drought. Glob. Change Biol. Vol. 22, No. 7: 2318–2328.
Schmid M. 2014: Abschätzung des Potenzials des Ägerisees für die Wärme- und Kältenutzung. Eawag, Kastanienbaum.
Schmid M., Köster O. 2016: Excess warming of a Central European lake driven by solar brightening. Water Resour. Res. Vol. 52, No. 10: 8103–8116.
Schmocker-Fackel P., Naef F. 2010: Changes in flood frequencies in Switzerland since 1500. Hydrol. Earth Syst. Sci. Vol. 14, No. 8: 1581–1594.
Schürch M., Bulgheroni M., Sinreich M. 2018: Température des eaux souterraines: un aperçu de l'état et de l'évolution en Suisse. Aqua Gas Vol. 7/8: 40–48.
Schwarzer C., Heinken T., Luthardt V., Joshi J. 2013: Latitudinal shifts in species interactions interfere with resistance of southern but not of northern bog-plant communities to experimental climate change. J. Ecol. Vol. 101, No. 6: 1484–1497.
Schwefel R., Gaudard A., Wüest A., Bouffard D. 2016: Effects of climate change on deep-water oxygen and winter mixing in a deep lake (Lake Geneva)–Comparing observational findings and modeling. Water Resour. Res. Vol. 52, No. 11: 8811–8826.
Sertić Perić M., Jolidon C., Uehlinger U., Robinson C.T. 2015: Long-term ecological patterns of alpine streams: An imprint of glacial legacies. Limnol. Oceanogr. Vol. 60, No. 3: 992–1007.
Seymour M., Deiner K., Altermatt F. 2016a: Scale and scope matter when explaining varying patterns of community diversity in riverine metacommunities. Basic Appl. Ecol. Vol. 17, No. 2: 134–144.
Seymour M., Seppälä K., Mächler E., Altermatt F. 2016b: Lessons from the macroinvertebrates: species-genetic diversity correlations highlight important dissimilar relationships. Freshw. Biol. Vol. 61, No. 11: 1819–1829.
Shama L.N.S., Robinson C.T. 2009: Microgeographic life history variation in an alpine caddisfly: plasticity in response to seasonal time constraints. Freshw. Biol. Vol. 54, No. 1: 150–164.
Sharpley A.N., McDowell R.W., Kleinman P.J.A. 2001: Phosphorus loss from land to water: integrating agricultural and environmental management. Plant Soil Vol. 237, No. 2: 287–307.
Short F.T., Kosten S., Morgan P.A., Malone S., Moore G.E. 2016: Impacts of climate change on submerged and emergent wetland plants. Aquat. Bot. Vol. 135: 3–17.

Shurin J.B., Clasen J.L., Greig H.S., Kratina P., Thompson P.L. 2012: Warming shifts top-down and bottom-up control of pond food web structure and function. Philos. Trans. R. Soc. B Biol. Sci. Vol. 367, No. 1605: 3008–3017.

Sievert N.A., Paukert C.P., Tsang Y.-P., Infante D. 2016: Development and assessment of indices to determine stream fish vulnerability to climate change and habitat alteration. Ecol. Indic. Vol. 67: 403–416.

Sinha R., Pearson L.A., Davis T.W., Burford M.A., Orr P.T., Neilan B.A. 2012: Increased incidence of Cylindrospermopsis raciborskii in temperate zones – Is climate change responsible? Water Res., Cyanobacteria: Impacts of climate change on occurrence, toxicity and water quality management Vol. 46, No. 5: 1408–1419.

Skjelkvåle B.L., Evans C., Larssen T., Hindar A., Raddum G.G. 2003: Recovery from acidification in European surface waters: A view to the future. AMBIO J. Hum. Environ. Vol. 32, No. 3: 170–175.

Sommaruga-Wögrath S., Koinig K.A., Schmidt R., Sommaruga R., Tessadri R., Psenner R. 1997: Temperature effects on the acidity of remote alpine lakes. Nature Vol. 387, No. 6628: 64–67.

Soria M., Leigh C., Datry T., Bini L.M., Bonada N. 2017: Biodiversity in perennial and intermittent rivers: a meta-analysis. Oikos Vol. 126, No. 8: 1078–1089.

Sorrell B.K., Mendelssohn I.A., McKee K.L., Woods R.A. 2000: Ecophysiology of Wetland Plant Roots: A Modelling Comparison of Aeration in Relation to Species Distribution. Ann. Bot. Vol. 86, No. 3: 675–685.

Stahl K., Weiler M., Kohn I., Freudiger D., Seibert J., Vis M., Gerlinger K., Böhm M. 2016: The snow and glacier melt components of streamflow of the river Rhine and its tributaries considering the influence of climate change. International Commission for the Hydrology of the Rhine Basin (KHR/CHR).

Steger C., Kotlarski S., Jonas T., Schär C. 2013: Alpine snow cover in a changing climate: a regional climate model perspective. Clim. Dyn. Vol. 41, No. 3–4: 735–754.

Steingruber S., Colombo L. 2006: Impacts of Air Pollution on Alpine Lakes and Rivers. Chemistry and biology in Alpine lakes and rivers in Southern Switzerland related to acidification from long-range transboundary air pollution: Monitoring results from 1980–2004. Environmental studies. Federal Office for the Environment, Bern, Switzerland.

Stomp M., Huisman J., Jongh F. de, Veraart A.J., Gerla D., Rijkeboer M., Ibelings B.W., Wollenzien U.I.A., Stal L.J. 2004: Adaptive divergence in pigment composition promotes phytoplankton biodiversity. Nature Vol. 432, No. 7013: 104.

Strahm I., Munz N., Leu C., Wittmer I., Stamm C. 2013: Landnutzung entlang des Gewässernetzes - Quellen für Mikroverunreinigungen. Aqua Gas No. 5: 36–44.

Straile D., Livingstone D.M., Weyhenmeyer G.A., George D.G. 2003: The Response of Freshwater Ecosystems to Climate Variability Associated with the North Atlantic Oscillation. Wiley Online Library.

Strayer D.L., Dudgeon D. 2010: Freshwater biodiversity conservation: recent progress and future challenges. J. North Am. Benthol. Soc. Vol. 29, No. 1: 344–358.

Streit K., Hagedorn F., Hiltbrunner D., Portmann M., Saurer M., Buchmann N., Wild B., Richter A., Wipf S., Siegwolf R.T.W. 2013: Soil warming alters microbial substrate use in alpine soils. Glob. Change Biol. Vol. 20, No. 4: 1327–1338.

Stucki P. 2010: Methoden zur Untersuchung und Beurteilung der Fliessgewässer: Makrozoobenthos Stufe F. Umwelt-Vollzug. Federal Office for the Environment, Bern.

Sukenik A., Quesada A., Salmaso N. 2015: Global expansion of toxic and non-toxic cyanobacteria: effect on ecosystem functioning. Biodivers. Conserv. Vol. 24, No. 4: 889–908.

Taylor R.G., Scanlon B., Döll P., Rodell M., van Beek R., Wada Y., Longuevergne L., Leblanc M., Famiglietti J.S., Edmunds M., Konikow L., Green T.R., Chen J., Taniguchi M., Bierkens M.F.P., MacDonald A., Fan Y., Maxwell R.M., Yechieli Y., Gurdak J.J., Allen D.M., Shamsudduha M., Hiscock K., Yeh P.J.-F., Holman I., Treidel H. 2013: Ground water and climate change. Nat. Clim. Change Vol. 3, No. 4: 322–329.

Tonolla D., Bruder A., Schweizer S. 2017: Evaluation of mitigation measures to reduce hydropeaking impacts on river ecosystems – a case study from the Swiss Alps. Sci. Total Environ. Vol. 574, No. Supplement C: 594–604.

Ulseth A.J., Bertuzzo E., Singer G.A., Schelker J., Battin T.J. 2018: Climate-Induced Changes in Spring Snowmelt Impact Ecosystem Metabolism and Carbon Fluxes in an Alpine Stream Network. Ecosystems Vol. 21, No. 2: 373–390.

Urban M.C., Bocedi G., Hendry A.P., Mihoub J.B., Pe'er G., Singer A., Bridle J.R., Crozier L.G., Meester L.D., Godsoe W., Gonzalez A., Hellmann J.J., Holt R.D., Huth A., Johst K., Krug C.B., Leadley P.W., Palmer S.C.F., Pantel J.H., Schmitz A., Zollner P.A., Travis J.M.J. 2016: Improving the forecast for biodiversity under climate change. Science Vol. 353, No. 6304.

Urrutia-Cordero P., Ekvall M.K., Ratcovich J., Soares M., Wilken S., Zhang H., Hansson L.-A. 2017: Phytoplankton diversity loss along a gradient of future warming and brownification in freshwater mesocosms. Freshw. Biol. Vol. 62, No. 11: 1869–1878.

Valk A.G. van der 1994: Effects of prolonged flooding on the distribution and biomass of emergent species along a freshwater wetland coenocline. Vegetatio Vol. 110, No. 2: 185–196.

Visco J.A., Apothéloz-Perret-Gentil L., Cordonier A., Esling P., Pillet L., Pawlowski J. 2015: Environmental Monitoring: Inferring the Diatom Index from Next-Generation Sequencing Data. Environ. Sci. Technol. Vol. 49, No. 13: 7597–7605.

Vonlanthen P., Bittner D., Hudson A.G., Young K.A., Müller R., Lundsgaard-Hansen B., Roy D., Piazza S.D., Largiader C.R., Seehausen O. 2012: Eutrophication causes speciation reversal in whitefish adaptive radiations. Nature Vol. 482, No. 7385: 357.

Vörösmarty C.J., McIntyre P.B., Gessner M.O., Dudgeon D., Prusevich A., Green P., Glidden S., Bunn S.E., Sullivan C.A., Liermann C.R., Davies P.M. 2010: Global threats to human water security and river biodiversity. Nature Vol. 467, No. 7315: 555–561.

VSA 2007: Abwassereinleitungen in Gewässer bei Regenwetter (STORM) - Richtlinie für die konzeptuelle Planung von Massnahmen.

Wahl B., Peeters F. 2014: Effect of climatic changes on stratification and deep-water renewal in Lake Constance assessed by sensitivity studies with a 3D hydrodynamic model. Limnol. Oceanogr. Vol. 59, No. 3: 1035–1052.

Walter F., Hänni E. 2018: Wege zu einer nachhaltigen Bodenpolitik. Nationales Forschungsprogramm NFP 68 "Nachhaltige Nutzung der Ressource Boden." Bern.

Walther G.-R. 2010: Community and ecosystem responses to recent climate change. Philos. Trans. R. Soc. Lond. B Biol. Sci. Vol. 365, No. 1549: 2019–2024.

Walther G.-R., Post E., Convey P., Menzel A., Parmesan C., Beebee T.J.C., Fromentin J.-M., Hoegh-Guldberg O., Bairlein F. 2002: Ecological responses to recent climate change. Nature Vol. 416, No. 6879: 389–395.

Walther G.-R., Roques A., Hulme P.E., Sykes M.T., Pyšek P., Kühn I., Zobel M., Bacher S., Botta-Dukát Z., Bugmann H., Czúcz B., Dauber J., Hickler T., Jarošík V., Kenis M., Klotz S., Minchin D., Moora M., Nentwig W., Ott J., Panov V.E., Reineking B., Robinet C., Semenchenko V., Solarz W., Thuiller W., Vilà M., Vohland K., Settele J. 2009: Alien species in a warmer world: risks and opportunities. Trends Ecol. Evol. Vol. 24: 686–693.

Wang B., Trabant D.C., Weller G., Anderson P. 1999: Effect of Climate Variability and Change on Water Resources in Alaska. Potential Consequences Clim. Var. Change Water Resour. U. S. Briane Adams Ed. Am. Water Works Assoc. Middleburg Va. TPS-99-1: 273–276.

Wedekind C., Küng C. 2010: Shift of Spawning Season and Effects of Climate Warming on Developmental Stages of a Grayling (Salmonidae). Conserv. Biol. Vol. 24 1418–1423.

Weyhenmeyer G.A., Jeppesen E., Adrian R., Arvola L., Blenckner T., Jankowski T., Jennings E., Nõges P., Nõges T., Straile D. 2007: Nitrate-depleted conditions on the increase in shallow northern European lakes. Limnol. Oceanogr. Vol. 52, No. 4: 1346–1353.

Whitehead P.G., Wilby R.L., Battarbee R.W., Kernan M., Wade A.J. 2009: A review of the potential impacts of climate change on surface water quality. Hydrol. Sci. J. Vol. 5: 101–123.

Whittaker R.J., Willis K.J., Field R. 2001: Scale and species richness: towards a general, hierarchical theory of species diversity. J. Biogeogr. Vol. 28, No. 4: 453–470.
Wigger F.W., Schmidlin L., Nagel P., von Fumetti S. 2015: Macroinvertebrate assemblages of natural springs along an altitudinal gradient in the Bernese Alps, Switzerland. Ann. Limnol. - Int. J. Limnol. Vol. 51, No. 3: 237–247.
Wilding N.A., White J.C., Chadd R.P., House A., Wood P.J. 2018: The influence of flow permanence and drying pattern on macroinvertebrate biomonitoring tools used in the assessment of riverine ecosystems. Ecol. Indic. Vol. 85: 548–555.
Winder M., Jassby A.D., MacNally R. 2011: Synergies between climate anomalies and hydrological modifications facilitate estuarine biotic invasions. Ecol. Lett. Vol. 14: 749–757.
Wood S.A., Borges H., Puddick J., Biessy L., Atalah J., Hawes I., Dietrich D.R., Hamilton D.P. 2017: Contrasting cyanobacterial communities and microcystin concentrations in summers with extreme weather events: insights into potential effects of climate change. Hydrobiologia Vol. 785, No. 1: 71–89.
Woodward F.I., Quaife T., Lomas M.R. 2010a: Praeger Review: Changing climate and the Irish landscape. Biol. Environ. Proc. R. Ir. Acad. Vol. 110B, No. 1: 1–16.
Woodward G., Perkins D.M., Brown L.E. 2010b: Climate change and freshwater ecosystems: impacts across multiple levels of organization. Philos. Trans. R. Soc. Lond. B Biol. Sci. Vol. 365, No. 1549: 2093–2106.
Woolway R.I., Simpson J.H., Spiby D., Feuchtmayr H., Powell B., Maberly S.C. 2018: Physical and chemical impacts of a major storm on a temperate lake: a taste of things to come? Clim. Change Vol. 151, No. 2: 333–347.
Wrona F.J., Prowse T.D., Reist J.D., Hobbie J.E., Lévesque L.M.J., Vincent W.F. 2006: Climate Change Effects on Aquatic Biota, Ecosystem Structure and Function. AMBIO J. Hum. Environ. Vol. 35, No. 7: 359–369.
Yankova Y., Neuenschwander S., Köster O., Posch T. 2017: Abrupt stop of deep water turnover with lake warming: Drastic consequences for algal primary producers. Sci. Rep. Vol. 7, No. 1: 13770.
Yvon-Durocher G., Allen A.P., Cellamare M., Dossena M., Gaston K.J., Leitao M., Montoya J.M., Reuman D.C., Woodward G., Trimmer M. 2015: Five Years of Experimental Warming Increases the Biodiversity and Productivity of Phytoplankton. PLOS Biol. Vol. 13, No. 12: e1002324.
Yvon-Durocher G., Hulatt C.J., Woodward G., Trimmer M. 2017: Long-term warming amplifies shifts in the carbon cycle of experimental ponds. Nat. Clim. Change Vol. 7, No. 3: nclimate3229.
Yvon-Durocher G., Montoya J.M., Trimmer M., Woodward G. 2011a: Warming alters the size spectrum and shifts the distribution of biomass in freshwater ecosystems. Glob. Change Biol. Vol. 17, No. 4: 1681–1694.
Yvon-Durocher G., Montoya J.M., Woodward G., Jones J.I., Trimmer M. 2011b: Warming increases the proportion of primary production emitted as methane from freshwater mesocosms. Glob. Change Biol. Vol. 17, No. 2: 1225–1234.
Zhang Y., Jeppesen E., Liu X., Qin B., Shi K., Zhou Y., Thomaz S.M., Deng J. 2017: Global loss of aquatic vegetation in lakes. Earth-Sci. Rev. Vol. 173: 259–265.
Zobrist J., Reichert P. 2006: Bayesian estimation of export coefficients from diffuse and point sources in Swiss watersheds. J. Hydrol. Vol. 329: 207–223.
Zobrist J., Schoenenberger U., Figura S., Hug S.J. 2018: Long-term trends in Swiss rivers sampled continuously over 39 years reflect changes in geochemical processes and pollution. Environ. Sci. Pollut. Res. Vol. 25, No. 17: 16788–16809.
Zubler E.M., Scherrer S.C., Croci-Maspoli M., Liniger M.A., Appenzeller C. 2014: Key climate indices in Switzerland; expected changes in a future climate. Clim. Change Vol. 123, No. 2: 255–271.